Puget Sound Science Update

Draft April 12, 2011

Puget Sound Partnership
our sound, our community, our chance

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Editor's note

The Puget Sound Science Update is a represents the state-of-the-science supporting the work of the Puget Sound Partnership to restore and protect the Puget Sound ecosystem. The Puget Sound Science Update represents an advancement in the development and use of science to support Puget Sound recovery in two important ways. First, the content of the Puget Sound Science Update was developed following a process modeled after the rigorous peer-review process used by the Intergovernmental Panel on Climate Change (IPCC), in which small author groups produced draft assessment reports synthesizing existing, peer-reviewed scientific information on specific topics identified by policy leaders. These drafts were peer-reviewed before the final reports were posted. Second, the Puget Sound Science Update will be published on-line following a collaborative model, in which further refinements and expansion occur via a moderated dialog using peer-reviewed information. Content eligible for inclusion must be peer-reviewed according to guidelines.

In the future, there will be two versions of the Update available at any time:

(1) a time-stamped document representing the latest peer-reviewed content (new time-stamped versions are likely to be posted every 4-6 months, depending on the rate at which new information is added); and

(2) a live, web-based version that is actively being revised and updated by users.

The initial Update you see here is a starting point to what we envision as an on-going process to synthesize scientific information about the lands, waters, and human social systems within the Puget Sound basin. As the document matures, it will become a comprehensive reporting and analysis of science related to the ecosystem-scale protection and restoration of Puget Sound. The Puget Sound Partnership has committed to using it as their 'one stop shopping' for scientific information—thus, it will be a key to ensuring that credible science is used transparently to guide strategic policy decisions.

The Update is comprised of four chapters, and you will note that some are still at earlier stages of completion than others. Over time—through the process of commissioned writing and user input through the web-based system—the content of all four chapters will be more deeply developed. We are relying in part on the scientific community to help ensure that the quality and nature of the scientific information contained in the Update meets the highest scientific standards.

Preface

Who are the authors of the Puget Sound Science Update?

Leading scientists formed teams to author individual chapters of the Puget Sound Science Update. These teams were selected by the Puget Sound Partnerhship's Science Panel in response to a request for proposals in mid-2009. Chapter authors are identified on the first page of each chapter. Please credit the chapter authors in citing the Puget Sound Science Update.

What are the Puget Sound Partnership and the Science Panel?

Please visit psp.wa.gov to learn about The Puget Sound Partnership.

Please visit science panel web page to learn about the Science Panel.

Has the Puget Sound Science Update been peer reviewed?

The original chapters of the Puget Sound Science Update were subjected to an anonymous peer review refereed by members of the Puget Sound Partnership's Science Panel. Reviewers are known only to referees on the Science Panel and the Partnership's science advisor.

What is "content pending review"?

The future web presentation is intended to offer a venue for updating, improving, and refining the material presented in the Puget Sound Science Update. Suggested amendments and additions are presented as "content pending review" on each page when an editor, perhaps working with a collaborating author, has developed some new content that has not yet been formally adopted for incorporation into the section. As "content pending review," this content should not be cited or should be cited in a way that makes clear that it is still in preparation.

How can I contribute new material to the Puget Sound Science Update?

Please visit the Puget Sound Partnership website to learn about how you can help improve, update, and refine the Puget Sound Science Update, or send an e-mail to pssu@psp.wa.gov to get the process started.

How can I cite the Puget Sound Science Update?

We recommend citations this version in the following format:

[Authors of specific chapter or section]. April 2011. [Section or chapter title] **in** Puget Sound Science Update, *April 2011* version. Accessed from http://www.psp.wa.gov/. Puget Sound Partnership. Tacoma, Washington.

"Content pending review" of the Puget Sound Science Update has not been fully reviewed for publication. If you elect to cite this information, we recommend that you contact the named author(s) to cite as a personal communication or cite the web-presentation using the following format:

[Authors of pending material]. In prep. Content pending review presented in *[Section or chapter title]* **in** Puget Sound Science Update. Accessed from http://www.psp.wa.gov/. Puget Sound Partnership. Tacoma, Washington.

Chapter 2A. The Biophysical Condition of Puget Sound

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Section 1. Introduction

Our objective in this section is to review the status and trends of biophysical components of Puget Sound that speak to the Puget Sound Partnerships key goals: species and food webs, habitats, water quality and water quantity. Each of these goals are multi-facetted, and a nearly limitless range of topics could be covered. Indeed, one of the qualities that make Puget Sound a natural treasure is the diversity of species and habitats that it supports. This diversity precludes detailed treatment of all ecosystem components and requires thoughtful selection of metrics that speak to ecological condition and policy goals.

An ideal process for selecting components would be a sequential approach allowing us to use the framework developed in Chapter 1 to evaluate multiple indicators followed by an analysis of data availability, status and trends therein. However, time constraints required that we work in parallel with the Chapter 1 effort, so our choice of focal components and our reporting is largely independent of that process. We do not use the term "indicators" when referring to these components because they have not been formally vetted as such.

Lacking a formal procedure or framework to select focal biophysical components, we adopted two overarching considerations in selecting components: metrics should be ecologically or policy relevant attributes of Puget Sound, and must have been the focus of sufficient study to permit status evaluation. Consequently, species that are recognized as important in the Puget Sound ecosystem, but for which sufficient data do not exist, were excluded from this analysis. Omissions based on data insufficiencies can be used to help guide decisions regarding data collection programs in the future. Additional guiding principles and considerations included the following: 1) culturally important species for which there are clear policy goals (e.g., harvested species, iconic species such as killer whales) were included whenever possible, along with critical species and habitats upon which they rely; 2) species of particular conservation concern were incorporated; 3) water quality and water quality components were chosen to reflect the topical emphasis of scientific study in each of those disciplines; 4) species that have been specifically identified as ecosystem indicators (via peer reviewed publications) were considered whenever possible.

This set of principles provided criteria that allowed a systematic approach to selection of components to include in this analysis. However, it did result in some noteworthy exclusions. For example, the status and trends of invasive species (e.g., *Spartina*, *Ciona*) are not reported. Analysis of zooplankton community composition and trends is limited by the paucity of data. Ocean acidification, a growing concern with potentially substantial impacts on shellfish aquaculture and natural communities, is not treated here. These and other omissions are not intended to imply that these are not important issues or components of the Puget Sound ecosystem, and we anticipate that the next iteration of the Puget Sound Science update can consider a broader range of metrics.

The ecosystem components treated in this chapter clearly emphasize marine and freshwater elements of the Puget Sound Watershed. This emphasis reflects the historical focus of the Puget Sound Science Update and the specific expertise of the lead authors. Even so, we selected terrestrial topics that have some linkage to aquatic portions of the watershed. We anticipate that future iterations of the Puget Sound Science Update will take a broader view and include many more terrestrial topics than we could incorporate in the present document.

There is a growing need for ecosystem assessments to guide ecosystem-based management. While the present evaluation might be considered a contribution to such an assessment, it is not an ecosystem assessment per se. Instead, it is an assessment of several ecosystem components. A full ecosystem assessment would also include a conceptual framework that links biological, physical and chemical processes and reports on key drivers and responses of each. Moreover, a quantitative synthesis of status and trends across all ecological and policy-relevant attributes of Puget Sound will provide a substantial advance.

Throughout, we aimed to vet available information to include only those results and conclusions that had undergone prior review. We recognized in advance that maintaining a requirement of peer-reviewed publication in scientific journals would be inappropriate: much of the scientific work on Puget Sound derives from long term monitoring that is not published in such journals. We therefore considered agency documents that were part of research reporting series to be sufficiently reviewed to be included in this chapter. This process revealed considerable differences among local agencies in the transparency of review processes for reports. There is a need for consistent standards and reporting practices among these agencies to permit an assessment of the thoroughness of reviews. We generally avoided citing previous iterations of the Puget Sound Science update as primary sources, because the nature and extent of review of components of those documents is also not clear. In some cases, monitoring data were used directly provided that the procedures used in collecting them had been reviewed and published.

Given these constraints, this chapter is not intended to be the final word on indicators for evaluating the status of Puget Sound. Indeed, Chapter 1 of the 2010 Puget Sound Science Update provides a substantial advance in improving the capacity to select ecologically meaningful indicators. Future versions of the Puget Sound Science Update will clearly benefit from the foundation that the present effort provides.

This chapter is organized primarily along the four Puget Sound Partnership goals, with separate sections for each ecosystem component. Within each summary, we provide background and rationale for inclusion in the Chapter, a brief treatment of threats and drivers to give the needed context. More thorough treatment of threats and drivers is provided in Chapter 3. We include in each section a synthesis of key data gaps and uncertainties. In some cases the uncertainties are scientific: uncertainties that can be resolved through additional scientific study. In other cases the uncertainties reflect emerging concepts, hypotheses and explanations that have not yet been vetted through a formal review process.

Species and Food Webs

1. Bivalves

Background

Molluscs in the Class Bivalvia feed on phytoplankton and detrital particles suspended in the water column, serving as a key trophic link between microscopic primary producers and higher consumers. Epibenthic bivalves can function as ecosystem engineers through the provision of hard substrate and three-dimensional biogenic structure, while infaunal bivalves can function as engineers through physical alteration of soft substrate habitats. Numerous native and non-native species of bivalves occur in Puget Sound, including important aquaculture species such as Pacific oysters (*Crassostrea gigas*), non-native invasive species such as the purple varnish clam (*Nutallia obscurata*), and species targeted in recreational fisheries (e.g., native littleneck clams and non-native Manila clams). The native geoduck clam, *Panopea generosa*, is valued as a commercially-fished species and as an aquaculture species. The native Olympia oyster, *Ostrea lurida*(also known as *Ostreola conchaphila*) currently is a restoration target in Puget Sound, having been depleted through human activities in the last century.

Geoduck clams

Geoducks are large Hiatellid clams distributed from Alaska to California. They can grow to shell lengths of 20 cm (Bureau et al. 2002), and are characterized by large fleshy siphons that can reach lengths of 1m. Geoducks are broadcast spawners with larval periods of 16 - 47 days (Goodwin and Pease 1989). After settlement, they exhibit limited mobility for 2-4 weeks, then burrow into the sand and begin feeding. Individuals are thought to reach maximum size within the first 10 years of life (Goodwin and Pease 1989), and can live for up to 168 years. Their longevity could render them particularly susceptible to over-exploitation (Orensanz et al. 2004).

In Puget Sound, geoducks occur primarily in low intertidal and subtidal habitats and are most abundant at depths of up to 20m, although observations of deeper individuals have been reported (Goodwin and Pease 1989). Found primarily in soft sediments consisting of sand and sand-mud, geoducks are contagiously distributed throughout the major basins of Puget Sound (Goodwin and Pease 1990). In a survey of 8,589 SCUBA transects, Goodwin and Pease (1990) found that geoduck abundance ranged from densities of 0 to 22.5 individuals/m2, with an average density of 1.7 individuals/m2. They found the highest densities in southern Puget Sound and in Hood Canal (Goodwin and Pease 1990).

Recreational and commercial fisheries for geoduck exist in Puget Sound. The recreational fishery typically occurs in intertidal habitats, while the commercial fishery occurs in subtidal habitats in areas leased from the State of Washington. Because the fishery is prosecuted in leased tracts, it is jointly managed by the Washington State Department of Natural Resources (WDNR) and the Washington Department of Fish and Game (WDFW). The current target for the commercial fishery in Puget Sound is 2.7% of the exploitable biomass based on a static value of 40% of the Maximum Sustainable Yield (MSY) (Bradbury et al. 2000). Recruitment of geoducks appears to be highly variable and driven by climatic forcing (Orensanz et al. 2004, Valero et al. 2004). Based on the combination of highly variable recruitment and long life span, Orensanz et al. (2004) caution that static exploitation targets may not be appropriate for this species. Geoduck abundance in Puget Sound is augmented through aquaculture, the ecological effects of which are not well understood (Feldmann et al. 2004, Straus et al. 2008).

Olympia oyster

As ecosystem engineers, oysters play an important role in the populations, communities and food webs where they occur (reviewed in Ruesink et al. 2005). Oyster beds provide structure and biogenic habitat for a suite of other invertebrates and fish (e.g., Lenihan et al. 2001). They also modify the physical and chemical properties of ambient water through feeding and excretion, maintaining high water clarity and conditions beneficial to macrophytes (Jackson et al. 2001, Ruesink et al. 2005).

The native Olympia oyster occurs from Alaska to Baja California, Mexico (Polson and Zacherl 2009). The size of the particles or phytoplankton ingested by oysters is determined by the size of their gills. Olympia oysters have larger gills and thus likely ingest larger particles than the common non-native Pacific oyster *Crassostrea gigas* (Couch and Hassler 1989). Olympia oysters are preyed upon by birds such as sea ducks and by crabs (Couch and Hassler 1989). They are relatively small, rarely reaching sizes greater than 5 cm, and have slow growth rates, typically reaching maturity after 4 years (Baker 1995, White et al. 2009b). Unlike many bivalves, fertilization is internal and larvae brood for 10-12 days within the mantle of females before spending 11-16 days as planktonic larvae (Dethier 2006). Olympia oyster spat have fairly narrow requirements for settlement, preferring hard, rugose substrates such as adult oyster shells (Trimble et al. 2009, White et al. 2009b). Beds of Olympia oysters are typically subtidal and individuals are known to be sensitive to extremes in temperature and desiccation stress (e.g., Baker 1995).

Status and Trends

Geoduck

Geoduck abundances for individual tracts throughout Puget Sound are estimated based on diver surveys conducted by WDFW according the methods described in Bradbury et al. (2000) and are posted online as part of the Geoduck Atlas , but abundances at the basin or sound-wide scales have not been summarized or published. Similarly, published fishery-independent population abundance data on trends in geoduck abundances are lacking.

Olympia Oyster

Olympia oysters in Washington state have been heavily exploited (Kirby 2004) and currently exist at abundances far lower than were reported historically (White et al. 2009a) (Figure 1). In Puget Sound, abundance was greatly reduced in the early 1900s despite the implementation of reserves throughout the Sound. Industrial pollution from paper mills is thought is thought to have contributed to the lack of effectiveness of the reserves (White et al. 2009a). The continued lack of population recovery is thought to be driven by a combination of limitations in the amount of preferred settlement substrate (adult conspecifics), competition with non-native oysters, and predation from introduced predators such as the Japanese drill *Ocinebrina inornata* (Buhle and Ruesink 2009, Trimble et al. 2009, White et al. 2009b). Their sensitivity to environmental extremes further restricts the habitats they can occupy (Trimble et al. 2009). Because of their low abundance, Olympia oysters currently are listed as a Washington State Candidate Species by WDFW . A number of projects for restoration of Olympia oyster populations have been initiated in Puget Sound (e.g., Brumbaugh and Coen 2009, Dinnel et al. 2009, White et al. 2009b).

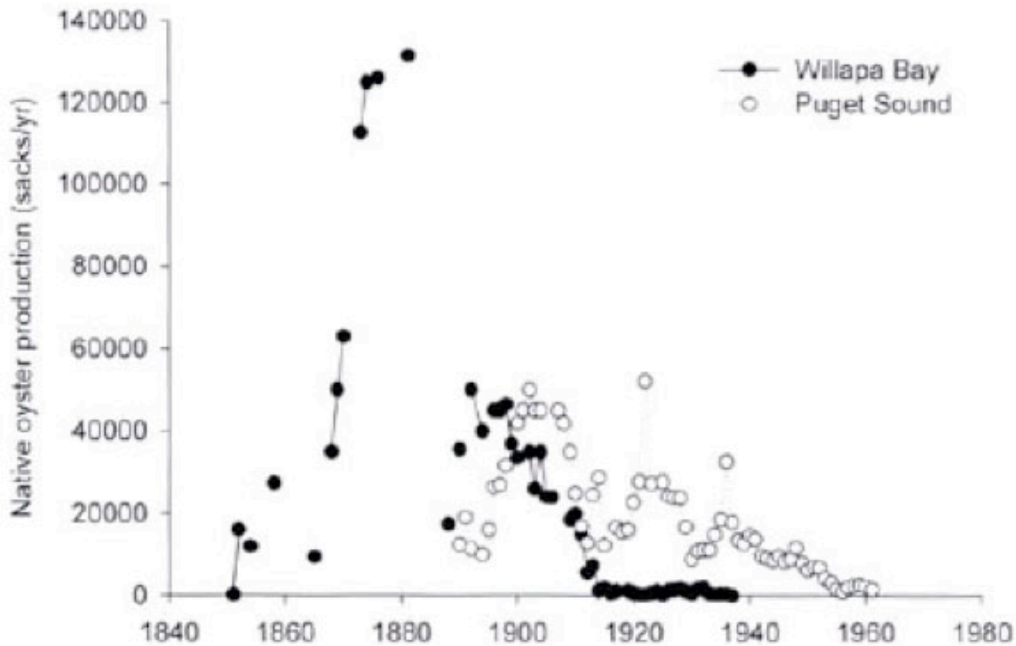


Figure 1. Olympia oyster harvest (1 sack is equal to approximately 4,000 individuals) in Willapa Bay (filled circles) and Puget Sound (open circles) from the mid 19th to mid 20th century based on Washington Marine Fish and Shellfish Landings (figure from White et al. 2009) (reprinted with permission from the Journal of Shellfish Research).

UNcertainties

There are several aspects of the current understanding of geoduck and Olympia oyster populations that are lacking. Geoduck tracts are surveyed frequently by WDFW yet estimates of basin and Sound-wide population status or trends have not been conducted. As such, spatial and temporal trends in geoduck abundances are not known for Puget Sound. Further, while cultivation of geoducks augments population abundances, the ecological effects of geoduck aquaculture practices in Puget Sound are not well understood (Feldmann et al. 2004, Straus et al. 2008). The sensitivity of Olympia oyster populations to abiotic stress and to predation from non-native predators pose challenges to the undertaking of restoring them to their former abundances and such the outcome of such efforts remains uncertain.

Summary

Native bivalves are essential components of the Puget Sound ecosystem. Geoduck clams are extremely long-lived, rendering them potentially susceptible to overexploitation. While geoduck abundance is estimated at small scales (tracts), published accounts of Sound-wide estimates of population status and trends are lacking. Abundances of Olympia oysters have been very low in Puget Sound since the 1940s, despite the fact that they are no longer targeted by fisheries. The importance of native oysters to ecosystems has prompted restoration efforts throughout Puget Sound.

Literature Cited

Baker, P. 1995. Review of ecology and fishery of the Olympia oyster, *Ostrea lurida*, with annotated bibliography. Journal of Shellfish Research 14:501-518.

Bradbury, A., B. Sizemore, D. Rothaus, and M. Ulrich. 2000. Stock assessment of subtidal geoduck clams (*Panopea abrupta*) in Washington. Washington Department of Fish and Wildlife, Olympia, WA.

Brumbaugh, R. D., and L. D. Coen. 2009. Contemporary approaches for small-scale oyster reef restoration to address substrate versus recruitment limitation: a review and comments relevant for the Olympia oyster, *Ostrea lurida* Carpenter 1864.(Report). Journal of Shellfish Research 28:147(115).

Buhle, E. R., and J. L. Ruesink. 2009. Impacts of invasive oyster drills on Olympia oyster (*Ostrea lurida* Carpenter 1864) recovery in Willapa Bay, Washington, United States.(Report). Journal of Shellfish Research 28:87(10).

Bureau, D., W. Hajas, N. W. Surry, C. M. Hand, G. Dovey, and A. Campbell. 2002. Age, size structure and growth parameters of geoducks (*Panopea abrupta* Conrad 1849) from 34 locations in British Columbia sampled between 1993 and 2000. Canadian Technical Report of Fisheries and Aquatic Sciences, Canada.

Couch, D., and T. J. Hassler. 1989. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest): Olympia Oyster. United States Fish and Wildlife Service Biological Report.

Dethier, M. 2006. Native Shellfish in Nearshore Ecosystems of Puget Sound.Puget Sound Nearshore Partnership Report No. 2006-04, Seattle, WA.

Dinnel, P. A., B. Peabody, and T. Peter-Contesse. 2009. Rebuilding Olympia oysters, *Ostrea lurida* Carpenter 1864, in Fidalgo Bay, Washington.(Report). Journal of Shellfish Research 28:79(77).

Feldmann, K., B. Vadopalas, D. Armstrong, C. Friedman, R. Hilborn, K. Naish, J. Orensanz, J. Ruesink, A. Suhrbier, A. Christy, D. Cheney, and J. Davis. 2004. Comprehensive literature review and synopsis of issues related to geoduck (*Panopea abrupta*) ecology and aquaculture production. Prepared for the Washington State Department of Natural Resources, Olympia, WA.

Goodwin, C. L., and B. C. Pease. 1990. Geoduck, Panopea abrupta (Conrad, 1849), size, density, and quality as related to various environmental parameters in Puget Sound, Washington. Journal of Shellfish Research 10:65-77.

Goodwin, L., and B. Pease. 1989. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest) -- Pacific geoduck clam. . United States Fish and Wildlife Service Biological Report, Brinnon, WA.

Jackson, J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, J. Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner, and R. R. Warner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293:629-637.

Kirby, M. X. 2004. Fishing down the coast: Historical expansion and collapse of oyster fisheries along continental margins. Proceedings of the National Academy of Sciences of the United States of America 101:13096-13099.

Lenihan, H. S., C. H. Peterson, J. E. Byers, J. H. Grabowski, G. W. Thayer, and D. R. Colby. 2001. Cascading of habitat degredation: Oyster reefs invaded by refugee fishes escaping stress. Ecological Applications 11:764-782.

Orensanz, J. M., C. M. Hand, A. M. Parma, J. Valero, and R. Hilborn. 2004. Precaution in the harvest of Methuselah's clams - the difficulty of getting timely feedback from slow-paced dynamics. Canadian Journal of Fisheries and Aquatic Sciences 61:1355-1372.

Polson, M. P., and D. C. Zacherl. 2009. Geographic distribution and intertidal population status for the Olympia oyster, *Ostrea lurida* Carpenter 1864, from Alaska to Baja.(Report). Journal of Shellfish Research 28:69(69).

Ruesink, J. L., H. S. Lenihan, A. C. Trimble, K. W. Heiman, F. Micheli, J. E. Byers, and M. C. Kay. 2005. Introduction of non-native oysters: Ecosystem effects and restoration implications. Annual Review of Ecology, Evolution, and Systematics 36:643-689.

Straus, K. M., L. M. Crosson, and B. V. Vadopalas. 2008. Effects of geoduck aquaculture on th environment: A synthesis of current knowledge. Washington Sea Grant Technical Report, Seattle, WA.

Trimble, A. C., J. L. Ruesink, and B. R. Dumbauld. 2009. Factors preventing the recovery of a historically overexploited shellfish species, *Ostrea lurida* Carpenter 1864.(Report). Journal of Shellfish Research 28:97(10).

Valero, J. L., C. Hand, J. M. Orensanz, A. M. Parma, D. Armstrong, and R. Hilborn. 2004. Geoduck (*Panopea abrupta*) recruitment in the Pacific Northwest: Long-term changes in relation to climate. California Cooperative Oceanic Fisheries Investigations Reports 45:80-86.

White, J., J. L. Ruesink, and A. C. Trimble. 2009a. The nearly forgotten oyster: *Ostrea lurida* Carpenter 1864 (Olympia oyster) history and management in Washington State.(Report). Journal of Shellfish Research 28:43(47).

White, J. M., E. R. Buhle, J. L. Ruesink, and A. C. Trimble. 2009b. Evaluation of Olympia oyster (*Ostrea lurida* carpenter 1864) status and restoration techniques in Puget sound, Washington, United States.(Report). Journal of Shellfish Research 28:107(106).

Pinto abalone

Background

Pinto abalone (*Haliotis kamtschatkana*) were once widely distributed throughout the waters of British Columbia and Washington state. In recent decades, populations have undergone sharp declines, likely in response to the combined stressors of overharvest, poaching, and sub-optimal environmental conditions (Campell 2000). Known for their large, muscular foot and their pearlescent oval shell, pinto abalone are slow-growing, long-lived marine snails and are typically found in nearshore rocky habitats in semi-exposed or exposed coastal regions. More than 60 abalone species are found worldwide but the pinto, or northern, abalone is the only species found in Washington State, where they range from Admiralty Inlet to the San Juan Islands and the Strait of Juan de Fuca and are typically found at depths to about 20 m (Bouma 2007).

Abalone are important herbivores in nearshore habitats, feeding primarily on drift macroalgae such as kelp and benthic diatom films. They can structure subtidal communities through the maintenance of substrata dominated by crustose coralline algae and through the facilitation of conspecific settlement. The larvae are planktonic and settle after approximately 7 -10 days in response to cues from both crustose coralline algae and from adults. Juvenile pinto abalone are cryptic until they reach a shell length of >50 mm.

Abalone are broadcast spawners. Consequently, the number and proximity of spawning adults determines the likelihood of successful fertilization (e.g., Babcock and Keesing 1999, Miner et al. 2006). At low population numbers, fertilization success may be low or nil, potentially limiting population recovery from overharvesting (Rothaus et al. 2008).

Status

The Washington Department of Fish and Wildlife (WDFW) regularly monitors the abundance of pinto abalone at 10 index stations throughout the San Juan Archipelago (Rothaus et al. 2008) (Figure 1). Because pinto abalone are highly patchy, cryptic and frequently associate with microhabitats such as rock crevices or patches of coralline algae that may themselves be patchily distributed, total abundances are not measured (Rothaus et al. 2008). Rather, repeated surveys at a system of index sites are conducted so that temporal trends in abalone abundance may be detected. The WDFW sites are composed mostly of bedrock and boulders encrusted with coralline algae, and support assemblages of kelp and other macroalgae (Rothaus et al. 2008). The sites range in size from 135 m2 to 380 m2, and individual animals are counted and measured during each survey.

Data from surveys made in 2006 showed an overall mean abalone density of 0.04 m-2 (Rothaus et al. 2008), which is well below the minimum densities for successful reproduction (0.15 individuals m-2) and recruitment (1 individual m-2) reported respectively by Babcock and Keesing (1999) and by Miner et al. (2006) for congeners of the pinto abalone.

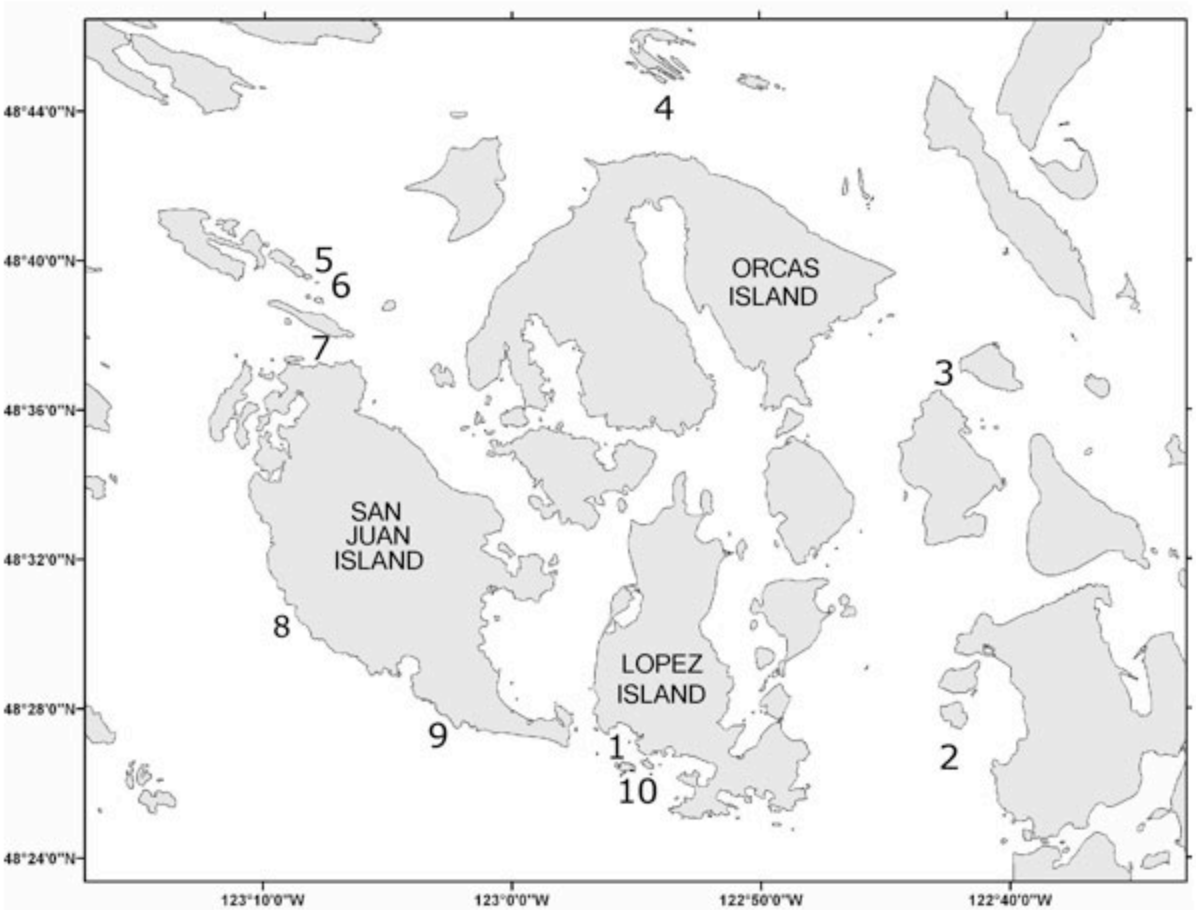


Figure 1. Map of WDFW *Haliotis kamtschatkana* index stations established in 1992 in the San Juan Archipelago, Washington State (Figure produced by WDFW and used with permission, methods according to Rothaus et al. 2008).

Trends

The decline of pinto abalone in Washington State has been of concern since the early 1990s (Rothaus et al. 2008). While commercial harvest of abalone has never been permitted in the state, the sport fishery may have extracted as many as 38, 200 individuals per year in the San Juan Archipelago (Bargmann 1984). It is therefore possible that abalone densities may have already been too low for successful fertilization or recruitment at the time of the sport fishery closure in 1994. WDFW listed the pinto abalone as a candidate species for protection in 1998 and NOAA Fisheries listed it as a federal species of concern in 2004. In 2008, WDFW identified pinto abalone as a Species of Greatest Conservation Need. In British Columbia, Canada, pinto abalone were uplisted to endangered in 2009, where populations are generally found at higher densities than Washington stocks (COSEWIC 2009).

The WDFW index site surveys in the San Juan Archipelago were repeated in 1994, 1996, 2003, 2004, 2005, 2006 and 2009. These surveys indicate a decline in abalone abundance of 83% from

1992 to 2009 (WDFW)(methods according to Rothaus et al. 2008)(Figure 2). Rothaus et al. (2008) also found an increase in mean shell length of 10.4 mm between 1992 and 2006, indicating a substantial shift in the size distribution of abalone populations, a pattern also present in the most recent survey in 2009(WDFW)(methods according to Rothaus et al. 2008)(Figure 3). This signifies a shift in abalone population age structure from younger to older animals, indicative of repeated recruitment failure (Rothaus et al. 2008). Recruitment failure following substantial declines in abalone density have been demonstrated elsewhere, for example in British Columbia, Canada (Tomascik and Holmes 2003) and in California (e.g., Miner et al. 2006). In Washington, the observed increases in mean shell length oppose the notion that the observed populations declines are a result of continued illegal harvest, because poaching is likely to result in a shift in length frequency toward smaller individuals (Rothaus et al. 2008). Pinto abalone populations may be unlikely to recover without intervention (Rothaus et al. 2008). Since 2004, a program of hatchery-based rearing and outplanting aimed at restoring abalone populations in Washington State has been led by the Puget Sound Restoration Fund (PSRF) and several local partners. In the summer of 2009, nearly 2,000 abalone were outplanted near Anacortes and Port Angeles, Washington.

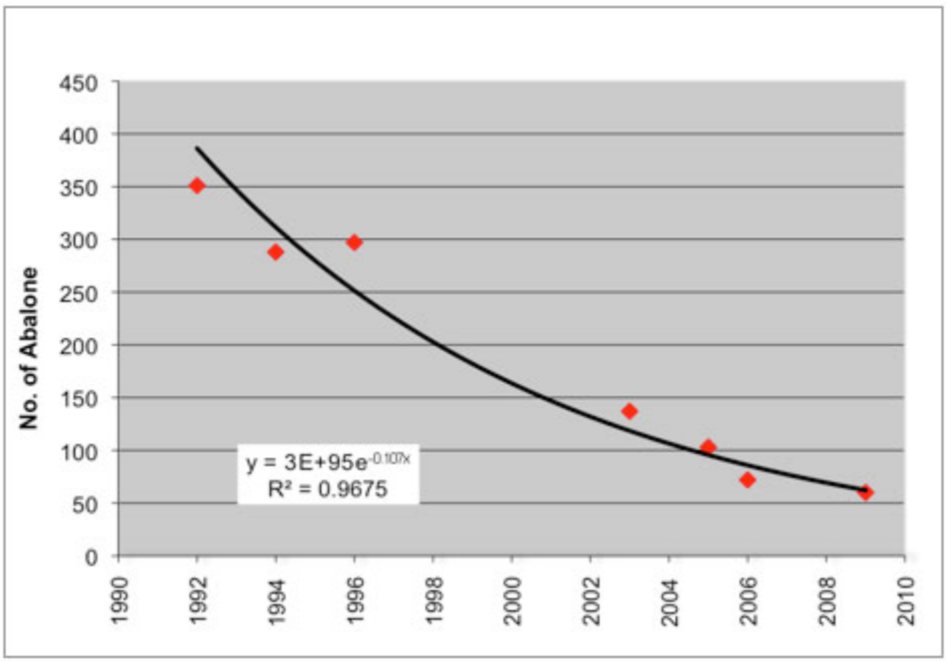


Figure 2. Pinto abalone abundance in the San Juan archipelago. Trends in abundance at 10 index stations from 1992 to 2009 (Figure produced by WDFW from unpublished data used with permission; methods according to Rothaus et al. 2008).

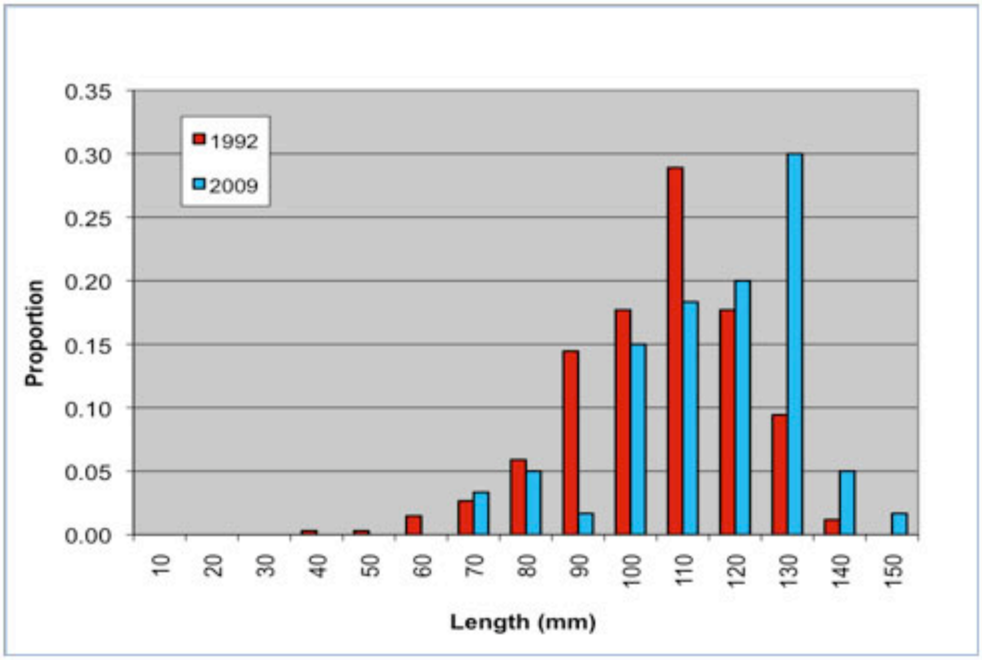


Figure 3. Pinto abalone shell length frequency in the San Juan archipelago. Trends in shell length from 10 index sites from 1992 to 2009 (Figure produced by WDFW from unpublished data used with permission; methods according to Rothaus et al. 2008).

Uncertainties

Many aspects of abalone biology and ecology are not well understood yet may be important in explaining both the decline and the recovery potential for pinto abalone in the Puget Sound region. While recreational fisheries likely played a role in the decline of pinto abalone in the San Juan Islands, the relative importance of harvesting and other factors is not known. While predation, habitat preferences, food availability and abiotic conditions will all likely affect the success of restoration efforts, the extent to which each of these factors may limit abalone populations is not well understood.

Summary

Pinto abalone are in severe decline in Puget Sound waters and are presently at densities where they may not be self-sustaining. Monitoring at index stations in the San Juan Islands showed an 83% decrease in abundance since 1992 despite their listing as federal species of concern, state candidate species, and the cessation of recreational harvest in 1994. Shell length surveys reveal that the population of pinto abalone in the San Juan Islands is aging without replacement although the direct causes of this recruitment failure warrant continued investigation. The long-term success of current hatchery-based rearing and outplanting programs is unknown at this time as efforts were recently initiated over the last five years.

Literature Cited

Babcock, R., and J. Keesing. 1999. Fertilization biology of the abalone *Haliotis laevigata*: laboratory and field studies. Canadian Journal of Fisheries and Aquatic Sciences 56:1668-1678.

Bargmann, G. G. 1984. Recreational diving in the State of Washington and the associated harvest of food fish and shellfish. State of Washington, Dept. of Fisheries, Marine Fish Program, Olympia, WA.

Bouma, J. V. 2007. Early life history dynamics of pinto abalone (*Haliotis kamtschatkana*) and implications for recovery in the San Juan archipelago, Washington State. Thesis (M S). School of Aquatic and Fisheries Science, University of Washington, Seattle, WA.

Campell, A. E. 2000. Workshop on rebuilding abalone stocks in British Columbia. Pages 173-185 in Can. Spec. Publ. Fish. Aquat. Sci. .

COSEWIC. 2009. COSEWIC assessment and update status report on the Northern Abalone *Haliotis kamtschatkana* in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, CAN.

Miner, C. M., J. M. Altstatt, P. T. Raimondi, and T. E. Minchinton. 2006. Recruitment failure and shifts in community structure following mass mortality limit recovery prospects of black abalone. Marine Ecology-Progress Series 327:107-117.

Rothaus, D. P., B. Vadopalas, and C. S. Friedman. 2008. Precipitous declines in pinto abalone (*Haliotis kamtschatkana kamtschatkana*) abundance in the San Juan Archipelago, Washington, USA, despite statewide fishery closure. Canadian Journal of Fisheries and Aquatic Sciences 65:2703(2709).

Dungeness Crabs

Background

Dungeness crabs (*Cancer magister*) occur throughout Washington waters, including the outer coast (mostly in coastal estuaries) and inland waters. Dungeness crabs use different habitats throughout their life cycle: as larvae they are planktonic, as juveniles they are found in intertidal mixed sand or gravel areas with algae or eelgrass (Holsman et al. 2006) and as adults they are found in subtidal or intertidal areas on sand, mud, or associated with eelgrass beds. Bare habitats are infrequently used by juveniles, most likely due to a lack of refuge from predation and decreased food abundance (McMillan et al. 1995). Vegetated, intertidal estuaries appear to be important nursery habitats for young crabs (Stevens and Armstrong 1984); older crabs have been shown to move progressively into unvegetated subtidal channels (Dinnel et al. 1986, Dethier 2006).

Annual settlement and survival of Dungeness crabs are typically variable. This variation stems from biotic factors such as predation and food availability, as well as abiotic factors such as water temperature and currents that transport larvae away from or toward nearshore areas. However, recruitment variability of Puget Sound populations is less than that seen in coastal populations (McMillan et al. 1995, Dethier 2006). There is evidence for local retention of Dungeness crab larvae within Puget Sound with a smaller proportion of recruits originating from coastal or oceanic stocks although this ratio is likely to vary from year to year (Dinnel et al. 1993, McMillan et al. 1995). Furthermore, the degree to which larvae originating in Puget Sound are transported through oceanic water before re-entering the sound is not well understood (Dethier 2006).

As predators and scavengers, Dungeness crabs feed upon a broad range of prey including small mollusks, crustaceans, clams, and fishes. They also prey for a wide variety of taxa, which varies with their life history stage. Larvae are preyed upon by coho and Chinook salmon and rockfishes; juveniles by a wide variety of fishes; and adults by fishes, seals, octopuses, and each other (generally when molting) (Orcutt et al. 1976, Reilly 1983, Dethier 2006).

Threats to Dungeness crabs include: low dissolved oxygen, variation in temperature and salinity, fisheries, habitat alteration or loss, and pollutants such as insecticides, hydrocarbons from oil spills and heavy metals. Because juvenile crabs rely on estuarine habitats and are also potentially more sensitive to toxins, early life history stages are likely to be more influenced by human activities (Dethier 2006).

StaTus

Due to their dependence on estuaries as juveniles, their value as recreational, commercial and tribal resources and their vulnerability to a suite of human impacts, Dungeness crab are included in the Washington Department of Fish and Wildlife (WDFW) Priority Habitats and Species List (Fisher and Velasquez 2008). However, there is currently no monitoring of Dungeness crab populations in Puget Sound that enable a reconstruction of population trends, status and sustainable harvest rates. Instead, time series of landings are used to gauge trends in population size over time. Commercial harvest quotas and recreational harvest season duration are determined from pre-season surveys that assess the relative abundance of mature females. The fishery is a male-only fishery, with a 6.25" (15.875 cm) carapace width minimum size. It is difficult to know whether temporally stable harvest rates represent stable population sizes or reflect changes in harvest effort or regulations (de Mutsert et al. 2008) Indeed, the increases in recreational landings may reflect increased fishing effort from a growing human population.

The current recommendations for Dungeness crab management in Puget Sound by WDFW include the reduction of habitat degradation by development, reduction in pollutants, and the reduction of impacts of fisheries (Fisher and Velasquez 2008)

Trends

Landings of Dungeness crab in Puget Sound have been highly variable, peaking at more than two million pounds in the late 1970s, declining in the 1980s, and rising again from the 1990s to 2005 (Dethier 2006). From 1995 to 2005, the biomass of Dungeness crab harvested annually by commercial, recreational, and tribal groups has shown an increase from six million pounds per season to approximately eight million pounds per season (Figure 1)(WDFW catch data, reported in Dethier 2006, PSP 2007) Increases in landings can reflect either an increase in fishing pressure or an increase in the abundance of the resource.

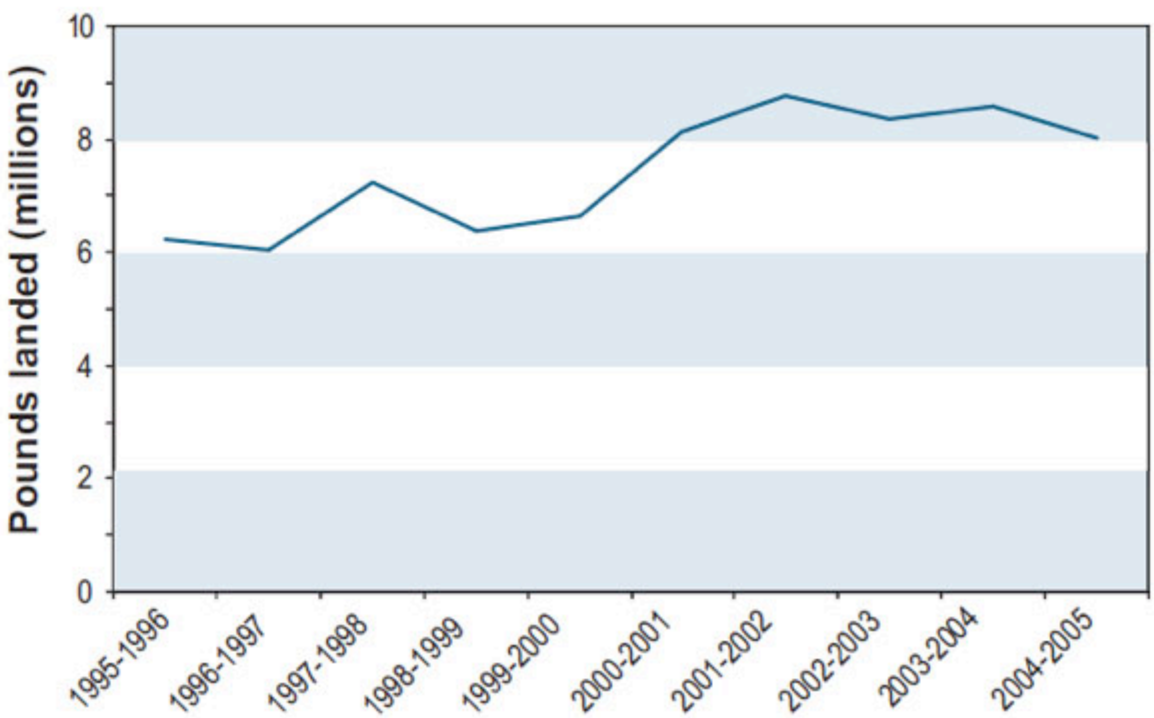


Figure 1. Dungeness crab harvest (commercial, recreational and tribal) landings from 1995 to 2005. (WDFW catch data, reported in Dethier 2006, PSP 2007) http://wdfw.wa.gov/fish/shelfish/crab/historic.htm).

Uncertainties

Because fisheries landings can be influenced by variables such as fishing effort that do not necessarily reflect crab population abundances, trends in landings data are not considered a reliable indicator of population status (de Mutsert et al. 2008). WDFW has estimated Dungeness crab abundance using a closed ring pot survey from 1999 to the present, however data from this survey have not been published.

Summary

Like many marine species with complex life histories, Dungeness crabs occupy different ecological niches throughout their life cycle and in therefore rely on multiple intact habitats. The associations between crabs and estuarine habitats, particularly nearshore habitats for juveniles may link habitat abundance and condition to the long-term health of Puget Sound Dungeness crabs. While landings data provide some information about the status of the fishery, they are not a reliable way to estimate natural population levels or trends.

Literature Cited

de Mutsert, K., J. H. Cowan, T. E. Essington, and R. Hilborn. 2008. Reanalyses of Gulf of Mexico fisheries data: Landings can be misleading in assessments of fisheries and fisheries ecosystems. Proceedings of the National Academy of Sciences 105:2740-2744.

Dethier, M. 2006. Native Shellfish in Nearshore Ecosystems of Puget Sound. Puget Sound Nearshore Partnership Report No. 2006-04, Seattle, WA.

Dinnel, P. A., D. A. Armstrong, and R. O. McMillan. 1993. Evidence for multiple recruitment-cohorts of Puget Sound Dungeness crab, Cancer magister. Marine Biology 115:53-63.

Dinnel, P. A., O. Russell, D. A. McMillan, D. A. Armstrong, T. C. Wainwright, A. J. Whiley, R. Burge, and R. Bumgarner. 1986. Padilla Bay Dungeness crab, *Cancer magister*, habitat study. Washington Department of Ecology, Seattle, WA.

Fisher, W., and D. Velasquez. 2008. Management recommendations for Washington's priority habitats and species: Dungeness crab (*Cancer magister*). Olympia, WA.

Holsman, K. K., P. S. McDonald, and D. A. Armstrong. 2006. Intertidal migration and habitat use by subadult Dungeness crab *Cancer magister* in a NE Pacific estuary. Marine Ecology-Progress Series 308:183-195.

McMillan, R. O., D. A. Armstrong, and P. A. Dinnel. 1995. Comparison of intertidal habitat use and growth-rates of 2 northern Puget Sound cohorts of 0+ age Dungeness crab, *Cancer magister*. Estuaries 18:390-398.

Orcutt, H. G., R. N. Tasto, P. W. Wild, C. W. Haugen, and E. E. Ebert. 1976. Dungeness crab research program. California Department of Fish and Game.

PSP. 2007. 2007 Puget Sound Update: Ninth Report of the Puget Sound Assessment and Monitoring Program. Olympia, WA.

Reilly, P. N. 1983. Predation on Dungeness crabs, *Cancer magister*, in Central California. California Department of Fish and Game, Sacramento, CA.

Stevens, B. G., and D. A. Armstrong. 1984. Distribution, abundance, and growth of juvenile dungeness crabs, *Cancer magister*, in Grays Harbor estuary, Washington. U.S National Marine Fisheries Service Fisheries Bulletin 82:469-483.

Jellyfish

Background

The term jellyfish is taxonomically broad, referring to gelatinous plankton in the phyla Ctenophora (comb jellies) and Cnidaria (all other jellyfish). While jellyfish have been components of pristine marine ecosystems for millennia, recent worldwide increases in the abundance of some jellyfish have been associated with anthropogenic perturbations such as eutrophication (Arai 2001), overfishing (Lynam et al. 2006), climate warming (Mills 2001, Lynam et al. 2004, Purcell 2005), and coastal development (Richardson et al. 2009). Because many jellyfish have a complex life history that includes free-living sexual and asexual phases, populations can increase rapidly when environmental conditions change to favor them.

Jellyfish blooms can disrupt human activities such as fishing, recreational beach use, and power plant operations (Purcell et al. 2007, Richardson et al. 2009). Moreover, jellyfish blooms can substantially alter food webs (e.g., Ruzicka et al. 2007, Pauly et al. 2009) by decreasing energy flow to higher trophic levels (Richardson et al. 2009) and by altering community composition of lower trophic levels through selective feeding (Purcell et al. 2007). Notably, the high degree of diet overlap between jellyfish and forage fish such as herring (Purcell and Arai 2001, Brodeur et al. 2008) is thought to be a driver of observed increases in jellyfish abundances in systems where forage fish are removed (Lynam et al. 2006). After such removals, fish recovery can be impeded by jellyfish predation on eggs and juvenile phases of their fish competitors (Purcell and Arai 2001), effectively preventing the reestablishment of fish populations (Lynam et al. 2006). Chum salmon (*Oncorhynchus keta*) are one of the few reported predators of jellyfish that occur in Puget Sound (Purcell and Arai 2001, Rice 2007)

Status

Data pertaining to jellyfish abundance in Puget Sound are scarce, but information is growing (Rice 2007, Reum et al. 2010). Biomass estimates determined from surface-towed trawl surveys conducted at 52 sites in Puget Sound in 2003 revealed relative abundances of jellyfish as high as 80% to 90% of the total trawl biomass at multiple sites in both the South Sound and in the Main Basin (Rice 2007)(Figure 1). By contrast, the observed relative abundances in the more northern regions of the Whidbey Basin and Rosario Strait were generally much lower (Figure 1). Importantly, when basin-wide data were considered, Rice (2007) noted an apparent inverse relationship between fish and jellyfish biomass. The jellyfish species observed were the Scyphomedusae *Cyanea capillata*, *Phacellophora camschatica*, *Aurelia* sp., the Hydromedusa *Aequorea* sp., and the Ctenophore *Pleurobrachia bachei* (Rice 2007). In June and September of 2007, Reum et al. (2010) conducted a more taxonomically-detailed study using bottom trawls in the northern and southern portions of Hood Canal (Hazel Point and Hoodsport, respectively) and in the Whidbey Basin (Useless Bay and Possession Sound). The species they reported were *Phacellophora camtschatica*, *Cyanea capillata*, *Aurelia labiata* and *Aequorea victoria*. While the abundances of jellyfish were both temporally and spatially variable, Reum et al. (2010) found that abundances were generally highest in June and at the southern portion of the Hood Canal mainstem near Hoodsport (Figure 2).

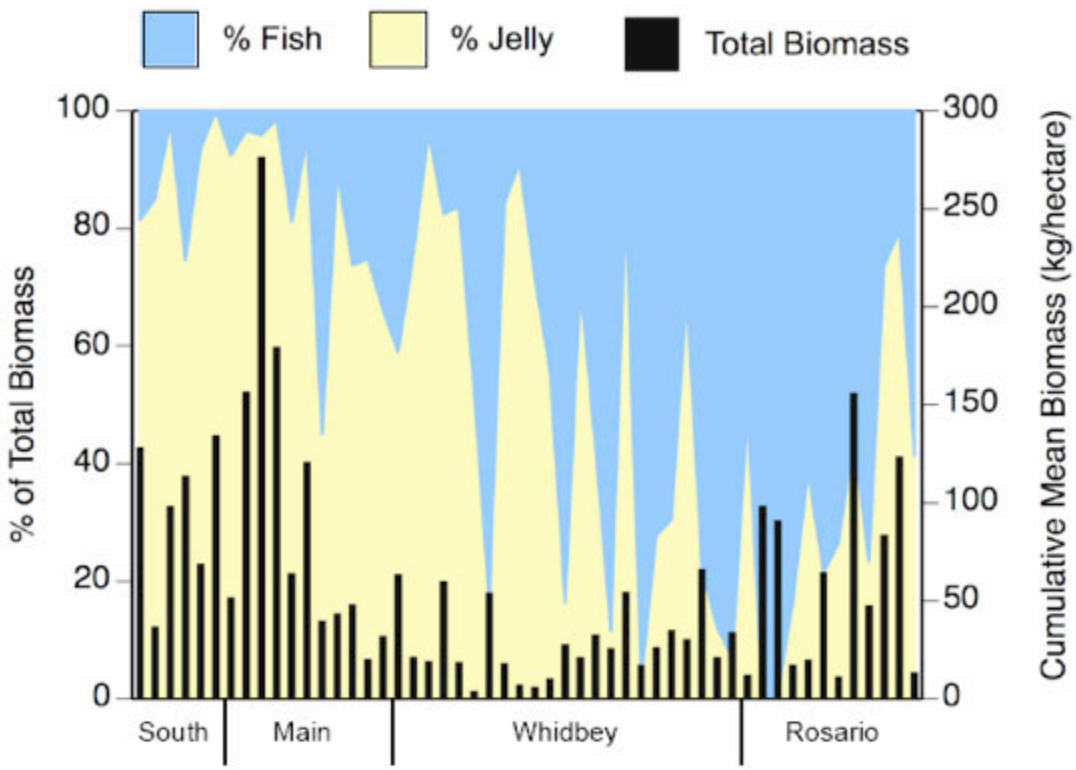


Figure 1. Percentage fish (blue area) and jelly (yellow area) in the total biomass (black bars) for sites within each region. Each bar is the sum of the four monthly means from May to August for each site. Reprinted with permission from Rice (2007).

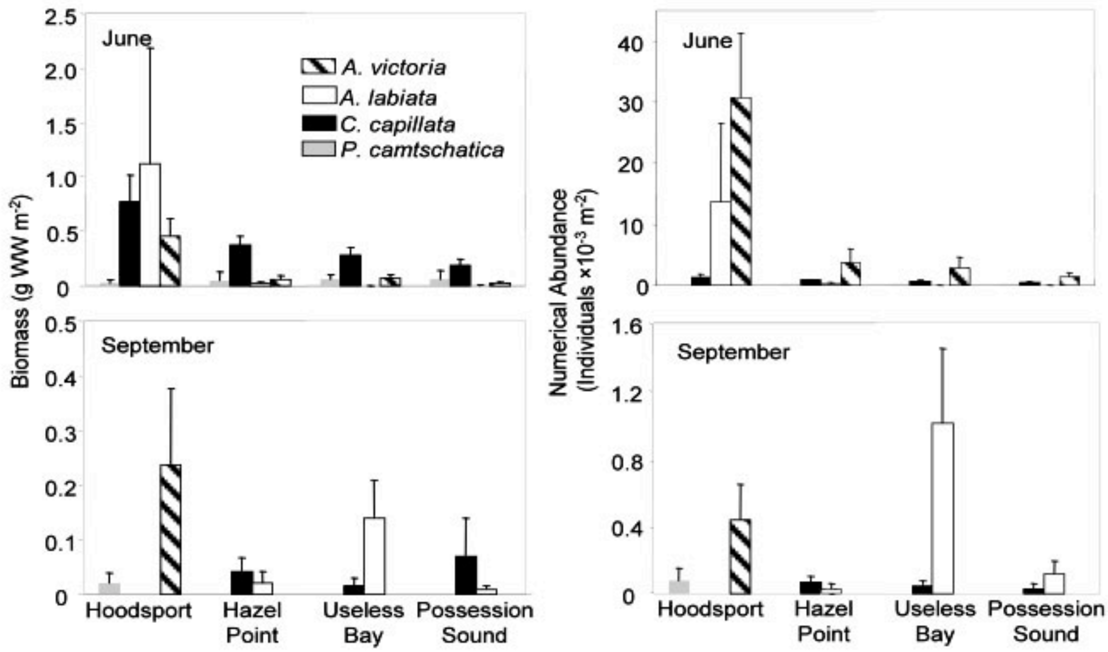


Figure 2. Biomass and numerical abundance densities sampled in June and September at four locations in Puget Sound, WA. Note that the y-axis for biomass and numerical abundances are scaled differently between June and September to better visualize variation in species composition. Error bars indicate standard deviation. Reprinted with permission from Northwest Science (Reum et al. 2010).

Trends

At this time it is not possible to determine temporal trends in jellyfish abundance in Puget Sound because existing data were collected using different methods and at different locations.

Uncertainties

The biology and ecology of most jellyfish are poorly known. In particular, knowledge of the asexually reproducing benthic polyp phase is limited (Boero et al. 2008). While it is clear from the limited available data that jellyfish are present in Puget Sound and that the likely causes of jellyfish outbursts (e.g., eutrophication, climate warming, coastal development and fishing pressure) also occur in Puget Sound to varying degrees, whether these factors are leading to increased jellyfish abundances has not been investigated. Because jellyfish have few predators, there is a high potential for them to disrupt food webs by displacing forage fish and other mid-trophic consumers, which could cause dramatic changes to the Puget Sound ecosystem. Indeed, a recent analysis of food webs in other temperate marine systems conducted by Samhouri et al. (2009) found that jellyfish were strongly correlated with multiple important ecosystem attributes, particularly those pertaining to trophic energy transfer.

Summary

While the direct mechanisms responsible for increases in jellyfish abundance in other marine systems are still being elucidated (Mills 2001, Purcell et al. 2007, Boero et al. 2008, Richardson et al. 2009), there appear to be associations between anthropogenically-perturbed systems and increased jellyfish abundance. The existing data are not sufficient to assess temporal patterns of jellyfish abundance in Puget Sound. Improved monitoring of spatial and temporal variability in jellyfish abundance as well as variation likely abiotic drivers would help to elucidate the causes and potential consequences of changing jellyfish abundance.

Literature Cited

Arai, M. N. 2001. Pelagic coelenterates and eutrophication: a review. Hydrobiologia 451:69-87.

Boero, F., J. Bouillon, C. Gravili, M. P. Miglietta, T. Parsons, and S. Piraino. 2008. Gelatinous plankton: irregularities rule the world (sometimes). Marine Ecology Progress Series 356:299-310.

Brodeur, R. D., C. L. Suchman, D. C. Reese, T. W. Miller, and E. A. Daly. 2008. Spatial overlap and trophic interactions between pelagic fish and large jellyfish in the northern California Current. Marine Biology (Berlin) 154:649-659.

Lynam, C. P., M. J. Gibbons, B. E. Axelsen, C. A. J. Sparks, J. Coetzee, B. G. Heywood, and A. S. Brierley. 2006. Jellyfish overtake fish in a heavily fished ecosystem. Current Biology 16:R492-R493.

Lynam, C. P., S. J. Hay, and A. S. Brierley. 2004. Interannual variability in abundance of North Sea jellyfish and links to the North Atlantic Oscillation. Limnology and Oceanography 49:637-643.

Mills, C. E. 2001. Jellyfish blooms: Are populations increasing globally in response to changing ocean conditions? Hydrobiologia:55-68.

Pauly, D., W. Graham, S. Libralato, L. Morissette, and M. Deng Palomares. 2009. Jellyfish in ecosystems, online databases, and ecosystem models. Hydrobiologia 616:67-85.

Purcell, J. E. 2005. Climate effects on formation of jellyfish and ctenophore blooms: a review. Journal of the Marine Biological Association of the UK 85:461-476.

Purcell, J. E., and M. N. Arai. 2001. Interactions of pelagic cnidarians and ctenophores with fish: a review. Hydrobiologia 451:27-44.

Purcell, J. E., S.-i. Uye, and W.-T. Lo. 2007. Anthropogenic causes of jellyfish blooms and their direct consequences for humans: a review. Marine Ecology Progress Series 350:153-174.

Reum, J., M. Hunsicker, and C. Paulsen. 2010. Species composition and relative abundance of large medusae in Puget Sound, WA. Northwest Science.84(2):131-140. doi: 10.3955/046.084.0202 .

Rice, C. A. 2007. Evaluating the biological condition of Puget Sound. Ph.D. dissertation. University of Washington, Aquatic and Fishery Sciences, Seattle, WA.

Richardson, A. J., A. Bakun, G. C. Hays, and M. J. Gibbons. 2009. The jellyfish joyride: causes, consequences and management responses to a more gelatinous future. Trends in Ecology & Evolution 24:312-322.

Ruzicka, J. J., R. D. Brodeur, and T. C. Wainwright. 2007. Seasonal food web models for the oregon inner-shelf ecosystem: Investigating the role of large jellyfish. California Cooperative Oceanic Fisheries Investigations Reports 48:106-128.

Samhouri, J., P. Levin, and C. Harvey. 2009. Quantitative evaluation of marine ecosystem indicator performance using food web models. Ecosystems 12:1283-1298.

Forage Fishes

Background

Forage fishes are small schooling fishes that form a critical link in the marine food web between zooplankton and larger fish and wildlife consumers. They occupy every marine and estuarine nearshore habitat in Washington, and much of the intertidal and shallow subtidal areas of the Puget Sound Basin are used by these species for spawning habitat. Status of forage fish populations can be an indicator of the health and productivity of nearshore systems (PSP 2009). Information on forage fish life history, distribution, and habitat preferences is summarized in Marine Forage Fishes of Puget Sound (Penttila 2007) and the Forage Fish Management Plan (Bargmann 1998).

The three most common forage fish species in the Puget Sound basin are Pacific herring (*Clupea pallasi*), surf smelt (*Hypomesus pretiosus*), and Pacific sand lance (*Ammodytes hexapterus*), and are therefore the focus of this section.

Pacific Herring

Pacific herring are a pelagic fish species found from northern Baja California to northern Honshu Island, Japan. They are found throughout the Puget Sound basin and are a mix of "resident" and "migratory" stocks (Gao et al. 2001, Penttila 2007, Stick and Lindquist 2009). Migratory populations cycle between the winter spawning grounds in the inside waters and the mouth of the Strait of Juan de Fuca in the summer, while resident stocks reside in the inside waters year-round (Penttila 2007). The faster individual growth rates observed in some herring populations are thought to be the result of fish leaving Puget Sound to feed in more productive oceanic waters and thus help to differentiate between migratory and resident stocks. For example, the Squaxin Pass herring population has a slower growth rate and is classified as "resident" while the Cherry Point population has a faster growth rate and is classified as "migratory" (Stick and Lindquist 2009).

Herring spawning occurs between January and April, with the majority of spawning taking place in February and March. Herring become ready to spawn over a two-month period by moving from deep water into shallow nearshore areas. The large natural and decadal oscillations in herring stock abundance are reflected in the area of spawning used annually. Most spawning areas appear to have "outlier" areas, used only during periods of high stock abundance, and "core" areas, used during periods of low stock abundance (Penttila 2007). Herring spawn on benthic marine macro-vegetation such as eelgrass or red macroalgae in the shallow subtidal and low intertidal region. Herring spawn preferentially in sheltered bays as opposed to vegetation beds on adjacent open shorelines (Stick and Lindquist 2009)(Figure 1).

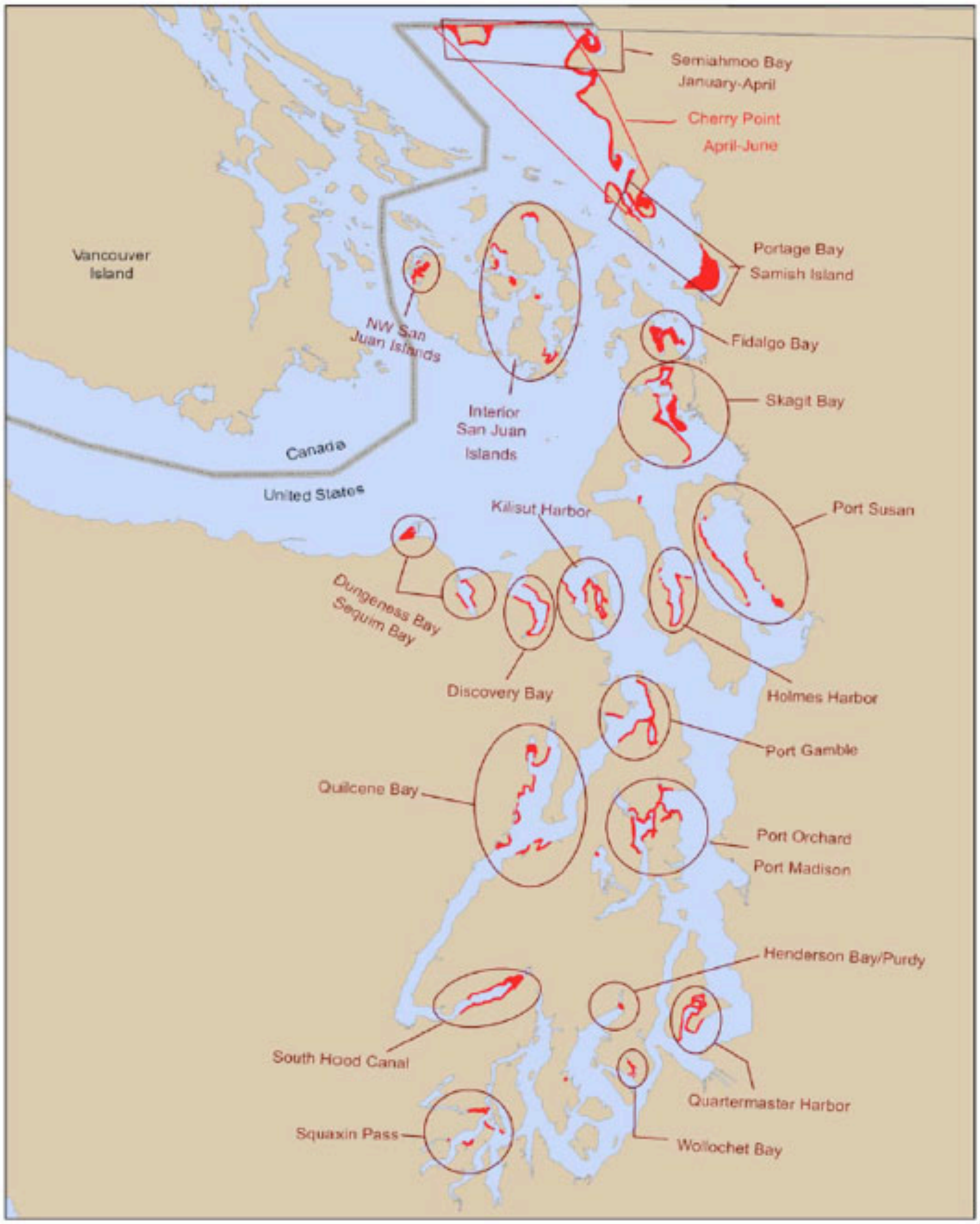


Figure 1. Documented Pacific herring spawning areas in Puget Sound (reprinted from Stick and Lindquist 2009 with permission from Washington Department of Fish and Wildlife).

Within the Puget Sound basin, autonomous stocks of herring are defined as having geographically distinct spawning areas and seasons. Two herring populations are deemed genetically distinct from other Puget Sound: the Cherry Point population which is distinctive for its late spawn timing (Small et al. 2005, Beacham et al. 2008, PSP 2008) and the Squaxin Pass population (Stick and Lindquist 2009)(Figure 1), which is thought to be spatially isolated from other populations (Small et al. 2005). Other sampled herring stocks show no evidence of genetic distinction (Small et al. 2005, Beacham et al. 2008), suggesting that these stocks may be part of a metapopulation where sufficient gene flow reduces genetic divergence (Stick and Lindquist 2009). If Puget Sound herring stocks act as a metapopulation, it may be more relevant to examine abundance trends on a larger scale than individual stock level, with Cherry Point and Squaxin Pass being the exceptions (Stick and Lindquist 2009).

Surf Smelt

Surf smelt are a nearshore species found from Long Beach, California to Chignik Lagoon, Alaska. They occur throughout the marine waters of Washington and in the southernmost region of Puget Sound. For the duration of their lifespan, surf smelt appear to inhabit shallow nearshore zones in the general area of their spawning (Penttila 2007).

Surf smelt spawning habitat is distributed throughout the Puget Sound basin and over a broad variety of conditions (e.g., variable salinity or shading). Spawning areas are usually occupied during summer (May-August), fall-winter (September-March), or year-round (monthly spawning with a seasonal peak)(Bargmann 1998, Penttila 2007). Spawning beaches are used on an annual basis, and as with Pacific herring, surf smelt have been shown to utilize “outlier” spawning sites during periods of high stock abundance (Penttila 2007).

Surf smelt use predictable shoreline areas for spawning across seasons; all spawning beaches first mapped by the Washington Department of Fish and Wildlife (WDFW) in the 1930s are still used by surf smelt. The critical habitat elements for spawning are substrate and tidal elevation. Surf smelt spawn in the uppermost one-third of the tidal range and most beaches appear suitable for surf smelt spawning habitat ranging from sheltered beaches to fully-exposed pebble beaches (Penttila 2007). Due to the diffuse nature of surf smelt spawning habitat there are no obvious grounds for stock definition in geographical terms.

Pacific Sand Lance

The Pacific sand lance occurs throughout the coastal northern Pacific Ocean from the Sea of Japan to southern California, and is widespread within the nearshore marine waters of Washington, including the entire Puget Sound basin. Sand lances inhabit nearshore waters and spawn between November and February. Sites and spawning habitats of sand lance are similar to that of surf smelt: upper intertidal sand and gravel beaches. Sand lance spawning often takes place on beaches at the distal ends of drift-cells, where accretionary shoreforms tend to occur. Because sand lance and surf smelt deposit eggs in the upper intertidal, they are particularly vulnerable to shoreline habitat modifications (Bargmann 1998).

Status

Of the forage fishes reviewed in this document, only Pacific herring populations have been monitored with sufficient detail to permit status evaluation. Surf smelt and sand lance populations are generally not considered threatened or endangered yet their abundances are currently unknown (Penttila 2007, PSP 2007).

Because of the dependence of forage fish on specific macro-vegetation for spawning, both environmental conditions and human activity (e.g., nearshore development) are likely to affect forage fish spawning biomass (Penttila 2007, Stick and Lindquist 2009). For this and other reasons (e.g., the difficulty in sampling adult populations), regulations have focused on managing forage fish spawning habitat. The Washington Administrative Code (WAC) (220-110), state Growth Management Act (GMA), and WDFW Priority Habitats and Species Program (PHS) all identify forage fish habitat as priority conservation "critical areas" or "areas of concern" for forage fish management (Penttila 2007).

Pacific Herring

There are 19 different stocks of Pacific herring in Puget Sound, based on timing and location of spawning activity (Bargmann 1998, PSP 2007). For 2007-2008, less than half of Puget Sound herring stocks were classified as healthy or moderately healthy (Stick and Lindquist 2009)(Table 1). This is similar to the status breakdown for the previous two-year periods (2003-04, 2005-06). The combined spawning biomass for all Puget Sound, excluding Cherry Point, is considered moderately healthy compared to the previous 25-year mean (11,656 tons for 2007-08 compared with 16,263 tons for 25-year mean). The abundance of south and central Puget Sound herring stocks, excluding Squaxin Pass (which is considered healthy at this time), are considered moderately healthy for 2007-08 (Stick and Lindquist 2009)(Table 1). The cumulative north Puget Sound regional spawning biomasses are considered depressed. Cherry Point continues to be considered critical; spawning biomass decreased during 2007 and 2008. Fidalgo Bay has also declined significantly since 1999 (Stick and Lindquist 2009)(Table 1). The Strait of Juan de Fuca regional status has generally been classified as critical, primarily due to Discovery Bay and Dungeness/Sequim Bay stocks suffering serious declines in biomass in recent years (Table 1) (Penttila 2007, PSP 2007, Stick and Lindquist 2009).

Table 1. Puget Sound herring stock status based on previous 2-year mean abundance compared to previous 25-year mean abundance (from Stick and Lindquist 2009).

STOCK STATUS - Describes a stock's current condition based primarily on recent (previous 2 year mean) abundance compared to long-term (previous 25 year mean) abundance.
Stock criteria such as survival, recruitment, age composition, and spawning ground habitat condition are also considered.
HEALTHY - A stock with recent two year mean abundance above or within 10% of the 25 year mean.
MODERATELY HEALTHY - A stock with recent two year mean abundance within 30% of the 25 year mean, and/or with high dependence on recruitment.
DEPRESSED - A stock with recent abundance well below the long term mean, but not so low that permanent damage to the stock is likely (i.e., recruitment failure).
CRITICAL - A stock with recent abundance so low that permanent damage to the stock is likely or has already occurred (i.e., recruitment failure).
DISAPPEARANCE - A stock which can no longer be found in a formerly consistently utilized spawning ground.
UNKNOWN - Insufficient assessment data to identify stock status with confidence.

Region / Stock	2008	2006	2004	2002	2000	1998	1996	1994
South-Central Puget Sound	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY
Squaxin Pass	HEALTHY	MOD. HEALTHY	HEALTHY	HEALTHY	HEALTHY	MOD. HEALTHY	MOD. HEALTHY	MOD. HEALTHY
Wollochet Bay	UNKNOWN	UNKNOWN	UNKNOWN	UNKNOWN				
Quartermaster Harbor	DEPRESSED	MOD. HEALTHY	MOD. HEALTHY	MOD. HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY
Port Orchard-Madison	HEALTHY	HEALTHY	MOD. HEALTHY	HEALTHY	HEALTHY	DEPRESSED	DEPRESSED	DEPRESSED
South Hood Canal	MOD. HEALTHY	HEALTHY	MOD. HEALTHY	MOD. HEALTHY	HEALTHY	MOD. HEALTHY	UNKNOWN	UNKNOWN
Quilcene Bay	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	UNKNOWN
Port Gamble	DEPRESSED	DEPRESSED	DEPRESSED	MOD. HEALTHY	HEALTHY	DEPRESSED	HEALTHY	HEALTHY
Kilisut Harbor	DEPRESSED	DEPRESSED	MOD. HEALTHY	HEALTHY	HEALTHY	MOD. HEALTHY	UNKNOWN	HEALTHY
Port Susan	MOD. HEALTHY	DEPRESSED	DEPRESSED	MOD. HEALTHY	MOD. HEALTHY	HEALTHY	DEPRESSED	MOD. HEALTHY
Holmes Harbor	HEALTHY	HEALTHY	HEALTHY	HEALTHY	DEPRESSED	HEALTHY	UNKNOWN	UNKNOWN
Skagit Bay	HEALTHY	HEALTHY	HEALTHY	HEALTHY	MOD. HEALTHY	MOD. HEALTHY	HEALTHY	UNKNOWN
North Puget Sound	DEPRESSED	DEPRESSED	DEPRESSED	DEPRESSED	DEPRESSED	DEPRESSED	MOD. HEALTHY	HEALTHY
Fidalgo Bay	DEPRESSED	DEPRESSED	DEPRESSED	HEALTHY	HEALTHY	HEALTHY	MOD. HEALTHY	MOD. HEALTHY
Samish/Portage Bay	HEALTHY	HEALTHY	MOD. HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	MOD. HEALTHY
Interior San Juan Is.	DEPRESSED	MOD. HEALTHY	DEPRESSED	MOD. HEALTHY	DEPRESSED	UNKNOWN	UNKNOWN	UNKNOWN
N.W. San Juan Is.	DISAPPEARANCE	DEPRESSED	CRITICAL	DEPRESSED	UNKNOWN	DEPRESSED	UNKNOWN	UNKNOWN
Semiahmoo Bay	MOD. HEALTHY	MOD. HEALTHY	DEPRESSED	MOD. HEALTHY	DEPRESSED	DEPRESSED	HEALTHY	HEALTHY
Cherry Point	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL	DEPRESSED	MOD. HEALTHY
Strait of Juan de Fuca	CRITICAL	DEPRESSED	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL
Discovery Bay	CRITICAL	DEPRESSED	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL
Dungeness/Sequim Bay	DEPRESSED	DEPRESSED	DEPRESSED	MOD. HEALTHY	HEALTHY	HEALTHY	HEALTHY	UNKNOWN
Puget Sound Combined	MOD. HEALTHY	HEALTHY	MOD. HEALTHY	HEALTHY	MOD. HEALTHY	MOD. HEALTHY	MOD. HEALTHY	HEALTHY
Individual Stock Comparison	**2008**	**2006**	**2004**	**2002**	**2000**	**1998**	**1996**	**1994**
HEALTHY	6 stocks	6 stocks	4 stocks	8 stocks	10 stocks	7 stocks	7 stocks	4 stocks
MOD. HEALTHY	3 stocks	4 stocks	5 stocks	7 stocks	2 stocks	3 stocks	2 stocks	5 stocks
DEPRESSED	6 stocks	7 stocks	6 stocks	1 stock	3 stocks	5 stocks	3 stocks	1 stock
CRITICAL	2 stocks	1 stock	3 stocks	2 stocks	2 stocks	2 stocks	1 stock	1 stock
DISAPPEARANCE	1 stock	0 stocks	0 stocks	0 stocks	0 stocks	0 stocks	0 stocks	0 stocks
UNKNOWN	1 stock	1 stock	1 stock	1 stock	1 stock	1 stock	5 stocks	7 stocks
	47% Healthy or Mod. Healthy	56% Healthy or Mod. Healthy	50% Healthy or Mod. Healthy	83% Healthy or Mod. Healthy	71% Healthy or Mod. Healthy	59% Healthy or Mod. Healthy	69% Healthy or Mod. Healthy	82% Healthy or Mod. Healthy

Trends

Pacific Herring

The cumulative spawning biomass of all Puget Sound herring stocks, except the Cherry Point stock, has fluctuated between about 10,000 to 16,000 tons (PSP 2009, Stick and Lindquist 2009) (Figure 2). Stocks in south and central Puget Sound have exhibited a general increasing trend, however this may be due to increased sampling effort since 1996. If the abundance of stocks are assumed to be at their mean levels during years when data are not available, then the estimated aggregate population sizes in the south and central Puget Sound stocks are comparable to those from 1970s and 1980s. Stocks in northern Puget Sound, excluding the Cherry Point stock, have remained at a low level of abundance (PSP 2009, Stick and Lindquist 2009) (Figure 2). Similarly, herring spawning biomass in the Strait of Juan de Fuca region continues to be very low and with the exception of 2006, the Discovery Bay herring stock has decreased steadily to between 200-250 tons annually since the mid 1990s (Stick and Lindquist 2009).

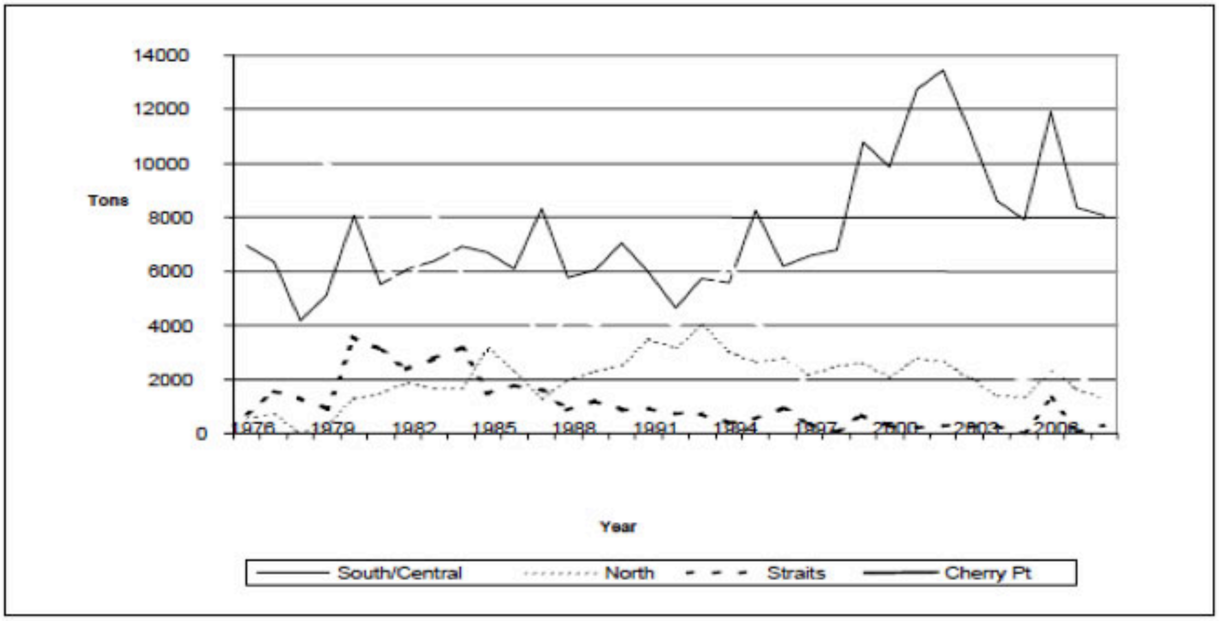


Figure 2. Estimated Puget Sound herring total spawning biomass by region and Cherry Point stock, 1976-2008 (data from Stick and Lindquist 2009, reprinted from PSP 2009).

Puget Sound herring stock abundance is significantly affected by mortality rates, which can be attributed to fishing and natural mortality (Stick and Lindquist 2009) (Figure 3). The mean estimated annual natural mortality rate for sampled Puget Sound herring stocks (excluding Cherry Point) since 1990 has averaged 72%, compared with typical mortality rates of 30-40% for herring worldwide. The Cherry Point herring stock annual mortality rate has increased to an average of 68% since 1990. Fishing mortality has averaged about 4% of estimated natural mortality since 1997. Predation, disease, and climatic changes are all potential causes of increased natural mortality (Stick and Lindquist 2009).

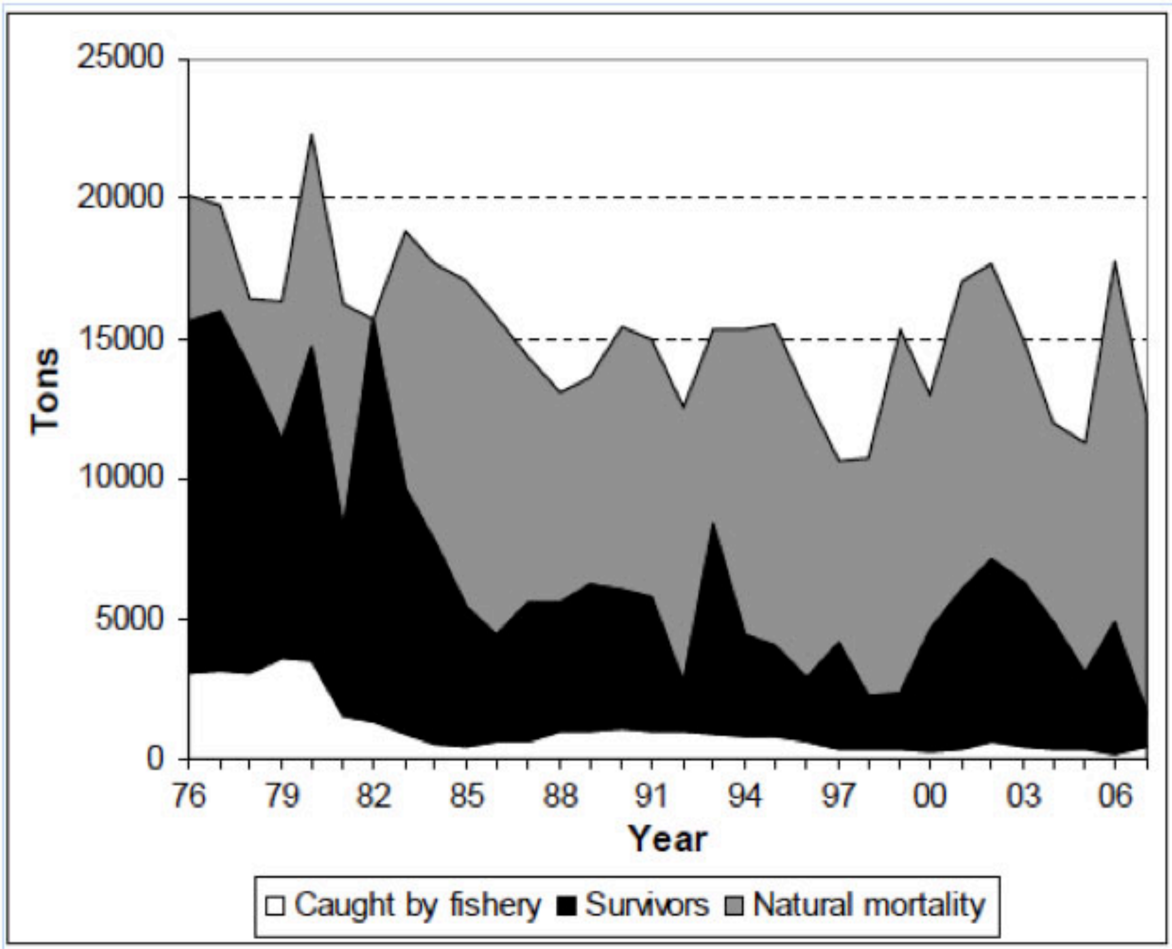


Figure 3. Annual tonnage estimates of herring in Puget Sound determined by natural mortality/survival rates, fishery harvest, and cumulative spawning biomass from 1976-2007 (reprinted from Stick and Lindquist 2009 with permission from Washington Department of Fish and Wildlife).

Uncertainties

Since the amount of data collected and the methods used for data collection differ across herring stocks and from year to year, Stick and Lindquist (2009) developed a system to evaluate the quality of the available information for each stock. They designated stocks which had a continuous time series of both acoustic-trawl and spawn deposition data as having "Good" data quality, stocks which had a continuous time series of only spawn deposition data as having "Fair" data quality, and populations for which there was an incomplete time series for either type of data as having "Poor" data quality. The majority of stocks assessed in this manner fell into the "Fair" category, with the best and most consistent data coming from Port Orchard/Madison and Cherry Point (Stick and Lindquist 2009)(Table 2).

Table 2. Puget Sound herring stock data quality determined by the amount of stock assessment data (evaluated in Stick and Lindquist 2009).

South/Central Puget Sound	Data Quality
Squaxin Pass	Fair
Wollochet Bay	Poor
Quartermaster Harbor	Fair
Port Orchard/Madison	Good
South Hood Canal	Poor
Quilcene Bay	Fair/Poor
Port Gamble	Fair
Kilisut Harbor	Fair/Poor
Port Susan	Fair
Holmes Harbor	Fair
Skagit Bay	Fair
North Puget Sound	
Fidalgo Bay	Fair
Samish/Portage Bay	Poor
Interior San Juan Islands	Poor
Northwest San Juan Island	Poor
Semiahmoo Bay	Fair
Cherry Point	Good
Strait of Juan de Fuca	
Discovery Bay	Fair
Dungeness/Sequim Bay	Poor

Good: A continuous time series of acoustic-trawl data & spawn deposition data.

Fair: A continuous time series of spawn deposition data only.

Poor: An incomplete time series of either type of stock assessment data.

Summary

Because of their reliance on near-shore habitats, the continued viability of these populations depends on the preservation of this habitat. Pacific Herring have a complicated population structure based on differences in the location and timing of spawning, although only two stocks are deemed genetically distinct. Data on population status are most extensive for Pacific Herring stocks, where current status and trends are mixed. The previously large Cherry Point stock is severely depressed from historical population levels. The prospect that this stock is now regulated by diseases has been raised and remains an active area of research. Long term assessment of other major species is needed to evaluate their current population levels and trends so that the impacts of habitat loss, fishing and climate change can be determined.

Literature Cited

Bargmann, G. 1998. Forage Fish Management Plan. Washington Department of Fish and Wildlife, Olympia, WA.

Beacham, T. D., J. F. Schweigert, C. MacConnachie, K. D. Le, and L. Flostrand. 2008. Use of microsatellites to determine population structure and migration of Pacific Herring in British Columbia and adjacent regions. Transactions of the American Fisheries Society 137:1795-1811.

Gao, Y. W., S. H. Joner, and G. G. Bargmann. 2001. Stable isotopic composition of otoliths in identification of spawning stocks of Pacific herring (*Clupea pallasi*) in Puget Sound. Canadian Journal of Fisheries and Aquatic Sciences 58:2113-2120.

Penttila, D. 2007. Marine Forage Fishes in Puget Sound. Puget Sound Nearshore Partnership Report No. 2007-03, Seattle, WA.

PSP. 2007. Puget Sound Update: Ninth Report of the Puget Sound Assessment and Monitoring Program. Page 260, Olympia, WA.

PSP. 2008. Species and Biodiversity Topic Forum. Puget Sound Partnership, Seattle, WA.

PSP. 2009. Ecosystem Status & Trends. Puget Sound Partnership, Seattle, WA.

Small, M. P., J. L. Loxterman, A. E. Frye, J. F. Von Bargen, C. Bowman, and S. F. Young. 2005. Temporal and spatial genetic structure among some Pacific Herring populations in Puget Sound and the Southern Strait of Georgia. Transactions of the American Fisheries Society 134:1329-1341.

Stick, K. C., and A. Lindquist. 2009. 2008 Washington State Herring Stock Status Report. Stock Status Report No. FPA 09-05, Olympia, WA.

Bentho-Pelagic Fish

Background

Bentho-pelagic fish utilize both demersal (bottom) habitats and shallower portion of the water column, often as part of diel migrations whereby fish feed in shallow water at night and move to deeper water to form schools during the day. Four currently or historically important species of bentho-pelagic fish in Puget Sound are the Pacific hake (*Merluccius productus*), the Pacific cod (*Gadus macrocephalus*), the Walleye pollock (*Theragra chalcogramma*) and the spiny dogfish (*Squalus acanthias*). Three of these species (Pacific hake, Pacific cod and Walleye pollock) were included in a petition for federal listing under the Endangered Species Act in 1999.

Pacific hake

Pacific hake (also known as Pacific whiting) form three spawning stocks in the Northeast Pacific: a coastal, highly migratory stock, a Strait of Georgia stock and a Puget Sound stock. Currently the two inland stocks and the coastal stock are federally recognized as Distinct Population Segments (DPS) based on genetic, demographic and behavioral differences (Gustafson et al. 2000), however more recent genetic evidence suggests further subdivision between southern Puget Sound and Strait of Georgia populations may be warranted (Iwamoto et al. 2004). In Puget Sound, Pacific hake form large seasonal spawning aggregations in Port Susan which was the target of a substantial fishery for many years (Pedersen 1985). Spawning activity has been also reported in Dabob Bay (Bailey and Yen 1983). Spawning in Puget Sound is thought to occur primarily from February to April (Gustafson et al. 2000). Pacific hake produce pelagic eggs which develop into larvae that feed primarily on copepods (McFarlane and Beamish 1985). As juvenile and small adults, the diet of hake is primarily euphausiid crustaceans which also undergo diel migrations (e.g., Mackas et al. 1997). Large adults consume a wide array of prey including amphipods, squid, Pacific herring, crabs, shrimp and juvenile Pacific hake (McFarlane and Beamish 1985, Gustafson et al. 2000). Pacific hake are also important prey for a suite of predators; this group includes walleye Pollock, Pacific cod, rockfish, spiny dogfish and marine mammals such as sea lions (McFarlane and Beamish 1985, Gustafson et al. 2000). Pacific hake in Puget Sound are believed to reach maturity at approximately 30 cm and 4-5 years of age; they can live for up to 20 years and reach sizes of 73 cm. The size at maturity and average body size of Pacific hake Puget Sound are reported to have decreased in Pacific hake from the 1980s to 2000 (WDFW data)(reported in Gustafson et al. 2000).

Pacific cod

Pacific cod occur in the Northeast Pacific occur from Alaska to California. Adult cod typically occupy deep habitats (50 – 300 m) and have been historically been observed forming spawning aggregations at multiple locations throughout Puget Sound (Palsson 1990, Gustafson et al. 2000). Although the review conducted by Gustafson et al. (2000) did not find conclusive evidence of population differentiation of North American Pacific cod, more recent otolith (Gao et al. 2005) and microsatellite (Cunningham et al. 2009) studies suggest that inland (Strait of Georgia and Puget Sound) populations are distinct from the coastal stocks. Pacific cod typically mature at 2-3 years of age at approximately 45 cm, with an estimated maximum lifespan of 18 years. Pacific cod occupy different habitats throughout their life cycle. Eggs are typically found in demersal habitats followed by a transition to the pelagic zone as larvae and small juveniles, settling to intertidal or subtidal sand or eelgrass habitats as large juveniles and moving to deep water as adults (reviewed by Gustafson et al. 2000). Juvenile cod feed on crustaceans such as shrimp, mysids and amphipods; the diet of adults is though to reflect the relative availability of prey with some preference for walleye pollock in large (>70 cm) adults (Gustafson et al. 2000). Pacific cod are preyed upon by a variety of predators including pelagic fishes, sea birds, whales, halibut, shark and other Pacific Cod.

Walleye pollock

Walleye pollock have a similar distribution to Pacific cod (from Alaska to California) and Puget Sound is thought to be one of the southernmost spawning locations across this range although this has been not well characterized (Gustafson et al. 2000). The degree of population structure of Pacific walleye pollock remains under investigation; earlier work using microsatellites did not find evidence of genetic structure (O'Reilly et al. 2004) whereas more recent work using non-neutral alleles has found evidence for differentiation between Puget Sound and other populations across its geographic range (Canino et al. 2005). Adult pollock are typically found between waters of 100 and 300 m depth and spawn at similar depths, with a lifespan of up to 17 years and a maximum size of up to 100 cm. Pollock eggs and larvae are pelagic, while juveniles and adults feed in surface waters at night and form schools in deeper water during the day although the presence of predators has been shown to shift this behavior to an association with structure such as seagrass (Sogard and Olla 1993). Larvae feed on copepod nauplii (e.g., Canino et al. 1991) while juveniles primarily feed euphausiids and other crustaceans (e.g., Brodeur 1998). Adult walleye pollock opportunistically feed on fishes, copepods and amphipods; a recent study of fish diets in Puget Sound found that walleye pollock stomachs contents were primarily pelagic invertebrates and small mobile benthic feeders (Reum and Essington 2008). Predators of walleye pollock include seabirds, marine mammals and other fish including cannibalistic interactions (summarized in Gustafson et al. 2000).

Spiny dogfish

Spiny dogfish are cartilaginous fish in the subclass Elasmobranchii along with sharks, skates and rays and are one of the longest-lived and latest-maturing taxa within this group, with an age-at-maturity of approximately 36 years and a lifespan of nearly 100 years (Saunders and McFarlane 1993). This life history strategy makes them particularly susceptible to overharvesting. While not typically harvested for consumption, they were intensely fished in the Puget Sound region in the 1940s for their oils, which are rich in Vitamin A. They are known to consume a variety of fish including cod and herring as well as crustaceans such as crabs (Jensen 1965). In the Northeast Pacific, a recent tagging study revealed them to be quite migratory, with some individuals utilizing habitats across British Columbia, the Strait of Georgia and Western Vancouver Island (McFarlane and King 2003).

Status

Pacific hake

As a result of declines in abundance in the Puget Sound population between the 1980s and late 1990s, the inland DPS (Strait of Georgia and Puget Sound) of Pacific hake is currently listed as a Federal Species of Concern and a Washington State Candidate Species (Palsson et al. 1998, Gustafson et al. 2000). If the Puget Sound population becomes recognized as a single DPS, the level of protection may increase. Commercial and recreational fisheries for hake in Puget Sound were closed in 1991 (Gustafson et al. 2000). Current population levels of Pacific hake in Puget Sound are not known.

Pacific cod

Pacific cod are currently listed as a Washington State Candidate Species (Palsson et al. 1998). Concerns over declines prompted the closure of the bottom trawl fishery near Port Townsend and Protection Island in 1991 and a prohibition of recreating fishing takes (Gustafson et al. 2000). However, as with Pacific hake, current population levels of Pacific cod in Puget Sound are not well known but are presumed to be low based on research survey (bottom trawl) and trap catch rates.

Walleye pollock

Walleye pollock, like Pacific cod, are listed as Washington State Candidate species (Palsson et al. 1998), yet current population levels of walleye pollock in Puget Sound have not been assessed. Daily recreational bag limits were reduced to zero in 1997 (Gustafson et al. 2000). Published reports that assess population status of Puget Sound walleye pollock are not available.

Spiny dogfish

Estimates of spiny dogfish population status in Puget Sound have not been reported in any peer-reviewed documents

Trends

Pacific hake

Pacific hake have undergone a decline in Puget Sound since the early 1980s. WDFW annual hydro-acoustic surveys combined with species composition and length distributions gathered from midwater trawls revealed an estimated 85 % decrease in the Port Susan total spawning biomass from 1983 to 1999 (WDFW data)(reported in Gustafson et al. 2000)(Figure 1). Trends for the Dabob Bay (Hood Canal) spawning population have not been documented (Gustafson et al. 2000).

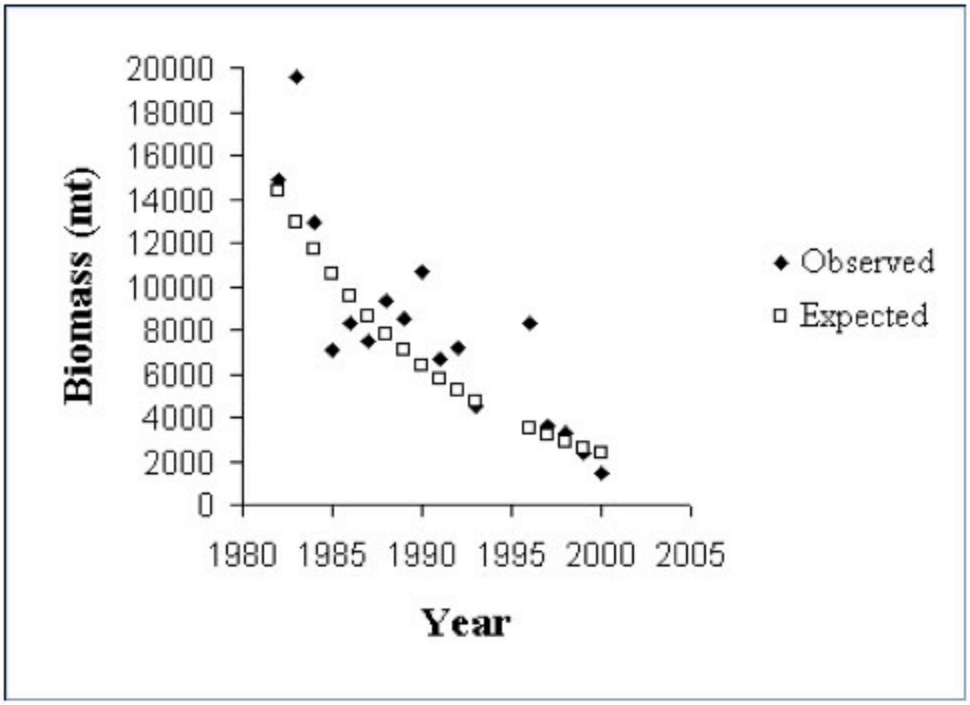


Figure 1. Results of a model predicting population declines (expected) and observed biomass of Pacific hake from WDFW trawls in Port Susan, Puget Sound from 1982 – 2000 (WDFW) (Reprinted from Gustafson et al. 2000; courtesy of NOAA Fisheries).

Pacific cod

The paucity of fishery-independent data on Pacific cod abundances makes population trends in Puget Sound difficult to assess, yet the decline in landings observed by WDFW and reported in Gustafson et al. (2000) combined with an apparent lack of subsequent reported occurrences suggest that populations in Puget Sound have likely declined substantially since the 1970s.

Walleye pollock

As with Pacific cod, the information available on walleye pollock abundance other than those based on fishery landings are lacking for Puget Sound. Fishing catches peaked in the late 1970s followed by a decline in the mid 1980s (WDFW data) (reported by Gustafson et al. 2000).

Spiny dogfish

Taylor and Gallucci (2009) report significant declines in spiny dogfish length and age at maturity and an increase in average fecundity between 1940–2000. However, these authors stressed it was difficult to discern whether these were due to density dependent effects following population declines or from climatic forcing.

Uncertainties

More information is needed to assess the current population sizes and future trends of all four key bentho-pelagic fish in Puget Sound. Specifically, analysis of long-term trends in abundance, population structure and dependence on environmental conditions is needed to ascertain status and key drivers.

Summary

Bentho-pelagic fish are important components of marine ecosystems and are often the targets of fishing pressure. In Puget Sound, Pacific hake, Pacific cod and walleye pollock were all once reported to be common and are now apparently much less abundant despite the fact that fishing pressure has been relieved. The direct causes for the declines and for the lack of rebounding are not well understood. All of these species are known to be susceptible to biophysical forcing and climatic regime shifts (Anderson and Piatt 1999, Hunt et al. 2002, Agostini et al. 2006, Agostini et al. 2008), and because Puget Sound cod and walleye pollock are at the southernmost extent of their range, these impacts may be particularly pronounced. Spiny dogfish, as an extremely long-lived shark has been shown to be susceptible to even low fishery pressure (Taylor and Gallucci 2009).

Literature Cited

Agostini, V. N., R. C. Francis, A. B. Hollowed, S. D. Pierce, C. Wilson, and A. N. Hendrix. 2006. The relationship between Pacific hake (*Merluccius productus*) distribution and poleward subsurface flow in the California Current System. Canadian Journal of Fisheries and Aquatic Sciences 63:2648-2659.

Agostini, V. N., A. N. Hendrix, A. B. Hollowed, C. D. Wilson, S. D. Pierce, and R. C. Francis. 2008. Climate-ocean variability and Pacific hake: A geostatistical modeling approach. Journal Of Marine Systems 71:237-248.

Anderson, P. J., and J. F. Piatt. 1999. Community reorganization in the Gulf of Alaska following ocean climate regime shift. Marine Ecology-Progress Series 189:117-123.

Bailey, K. M., and J. Yen. 1983. Predation by a carnivorous marine copepod, Euchaeta elongata Esterly, on eggs and larvae of the Pacific hake, *Merluccius productus*. J. Plankton Res. 5:71-82.

Brodeur, R. 1998. Prey selection by age-0 walleye pollock, *Theragra chalcogramma*, in nearshore waters of the Gulf of Alaska. Environmental Biology of Fishes 51:175-186.

Canino, M. F., K. M. Bailey, and L. S. Incze. 1991. Temporal and geographic differences in feeding and nutritional condition of walley pollock larvae *Theragra chalcogramma* in Shelikof Strait, Gulf of Alaska T. Marine Ecology-Progress Series 79:27-35.

Canino, M. F., P. T. O'Reilly, L. Hauser, and P. Bentzen. 2005. Genetic differentiation in walleye pollock (*Theragra chalcogramma*) in response to selection at the pantophysin (PanI) locus. Canadian Journal of Fisheries and Aquatic Sciences 62:2519-2529.

Cunningham, K. M., M. F. Canino, I.F.Spies, and L. Hauser. 2009. Genetic isolation by distance and localized fjord population structure in Pacific cod (*Gadus macrocephalus*): limited effective dispersal in the northeastern Pacific Ocean. Canadian Journal of Fisheries and Aquatic Sciences 66:153-166.

Gao, Y., G. Bargmann, U. Brand, and D. Noakes. 2005. Stable Isotopic and trace elemental compositions of otoliths and the stock structure of Pacific Cod, *Gadus macrocephalus*. Environmental Biology of Fishes 74:335-348.

Gustafson, R. G., W. H. Lenarz, B. B. McCain, C. C. Schmitt, W. S. Grant, T. L. Builder, and R. D. Methot. 2000. Status review of the Pacific Hake, Pacific Cod, and Walleye Polluck from Puget Sound, WA. NOAA, Department of Commerce.

Hunt, G. L., P. Stabeno, G. Walters, E. Sinclair, R. D. Brodeur, J. M. Napp, and N. A. Bond. 2002. Climate change and control of the southeastern Bering Sea pelagic ecosystem. Deep-Sea Research II 49:5821-5823.

Iwamoto, E., M. Ford, and R. Gustafson. 2004. Genetic Population Structure of Pacific Hake, *Merluccius productus*, in the Pacific Northwest. Environmental Biology of Fishes 69:187-199.

Jensen, A. L. 1965. Life history of the spiny dogfish. Fishery Bulletin 65:527-554.

Mackas, D. L., R. Kieser, M. Saunders, D. R. Yelland, R. M. Brown, and D. F. Moore. 1997. Aggregation of euphausiids and Pacific Lake (*Merluccius productus*) along the outer continental shelf off Vancouver Island. Canadian Journal of Fisheries and Aquatic Sciences 54:2080-2096.

McFarlane, G. A., and R. J. Beamish. 1985. Biology and fishery of Pacific whiting, *Merluccius productus* in the Strait of Georgia Marine Fisheries Review 47:23-34.

McFarlane, G. A., and J. R. King. 2003. Migration patterns of spiny dogfish (*Squalus acanthias*) in the North Pacific Ocean. Fishery Bulletin 101:358-367.

O'Reilly, P. T., M. F. Canino, K. M. Bailey, and P. Bentzen. 2004. Inverse relationship between F-ST and microsatellite polymorphism in the marine fish, walleye pollock (*Theragra chalcogramma*): implications for resolving weak population structure. Molecular Ecology 13:1799-1814.

Palsson, W. A. 1990. Pacific cod (*Gadus macrocephalus*) in Puget Sound and adjacent water: biology and stock assessment. Washington Department of Fish and Wildlife, Olyumpia, WA.

Palsson, W. A., T. J. Northup, and M. W. Barker. 1998. Puget Sound groundfish management plan (revised). Washington Department of Fish and Wildlife, Olyumpia, WA.

Pedersen, M. 1985. Puget Sound Pacific whiting, *Merluccius productus*, resource and industry: An overview. Marine Fisheries Review 47:35-38.

Reum, J., and T. Essington. 2008. Seasonal variation in guild structure of the Puget Sound demersal fish community. Estuaries and Coasts 31:790-801.

Saunders, M. W., and G. A. McFarlane. 1993. Age and length at maturity of the female spiny dogfish, *Squalus acanthias*, in the Strait of Georgia, British Columbia, Canada. Environmental Biology of Fishes 38:49-57.

Sogard, S., and B. Olla. 1993. The influence of predator presence on utilization of artificial seagrass habitats by juvenile walleye pollock, *Theragra chalcogramma*. Environmental Biology of Fishes 37:57-65.

Taylor, I. G., and V. F. Gallucci. 2009. Unconfounding the effects of climate and density dependence using 60 years of data on spiny dogfish (*Squalus acanthias*). Canadian Journal of Fisheries and Aquatic Sciences 66:351-366.

Rockfish

Background

Rockfish are bony fish in the Scorpaenid family, primarily in the genus *Sebastes*. Approximately 28 species of rockfish are reported from Puget Sound (Palsson et al. 2009), spanning a range of life-history types, habitats, and ecological niches. This diversity makes rockfish challenging to manage as a group and consequently, single-species management approaches have been recommended (Musick et al. 2000, Parker et al. 2000, Stout et al. 2001, Palsson et al. 2009, WDFW 2009). Rockfish in Pacific waters are among the most long-lived of teleost fishes and have low average annual reproductive success (Love et al. 2002). In combination, these characteristics make rockfish particularly susceptible to over-fishing. All of the rockfish in Puget Sound are classified as having Low or Very Low productivity according to definitions specified by the American Fisheries Society (AFS) (Musick 1999, Musick et al. 2000).

Rockfish have a biphasic life history in which pelagic larvae spend 1-2 months in the water column followed by settlement to benthic habitats that shift over ontogeny. In Puget Sound, settling rockfish are thought to associate with a variety of habitats including eelgrass, kelp, drift vegetation, and cobble fields, while many species as adults are found associated with deeper, high-relief rocky substrates (Palsson et al. 2009). While diet varies with species, developmental stage and location within the Sound, primary prey items for rockfish include Pacific herring, crabs, shrimp, surfperch, greenlings, and benthic invertebrates such as amphipods (reviewed by Palsson et al. 2009). In turn, rockfish, particularly as juveniles, are preyed upon by suite of predators including lingcod (Beaudreau and Essington 2007), salmonids and other fish (Palsson et al. 2009), while adults have been shown to be consumed by marine mammals such as harbor seals (Lance and Jeffries 2007).

Although rockfish larvae are pelagic, there is genetic evidence for limited dispersal within Puget Sound for the quillback (*S. maliger*) and copper (*S. caurinus*) rockfish (Seeb 1998) as well as for differentiation from coastal populations of brown rockfish (*S. auriculatus*) (Buonaccorsi et al. 2002). This degree of population structure is consistent with other genetic and otolith studies from coastal Pacific rockfish populations (Cope 2004, Miller et al. 2005, Burford 2009). Because of these findings, populations of each species of rockfish in the northern and southern portions of Puget Sound are recognized by WDFW to be separate stocks (Palsson et al. 2009) (Figure 1).

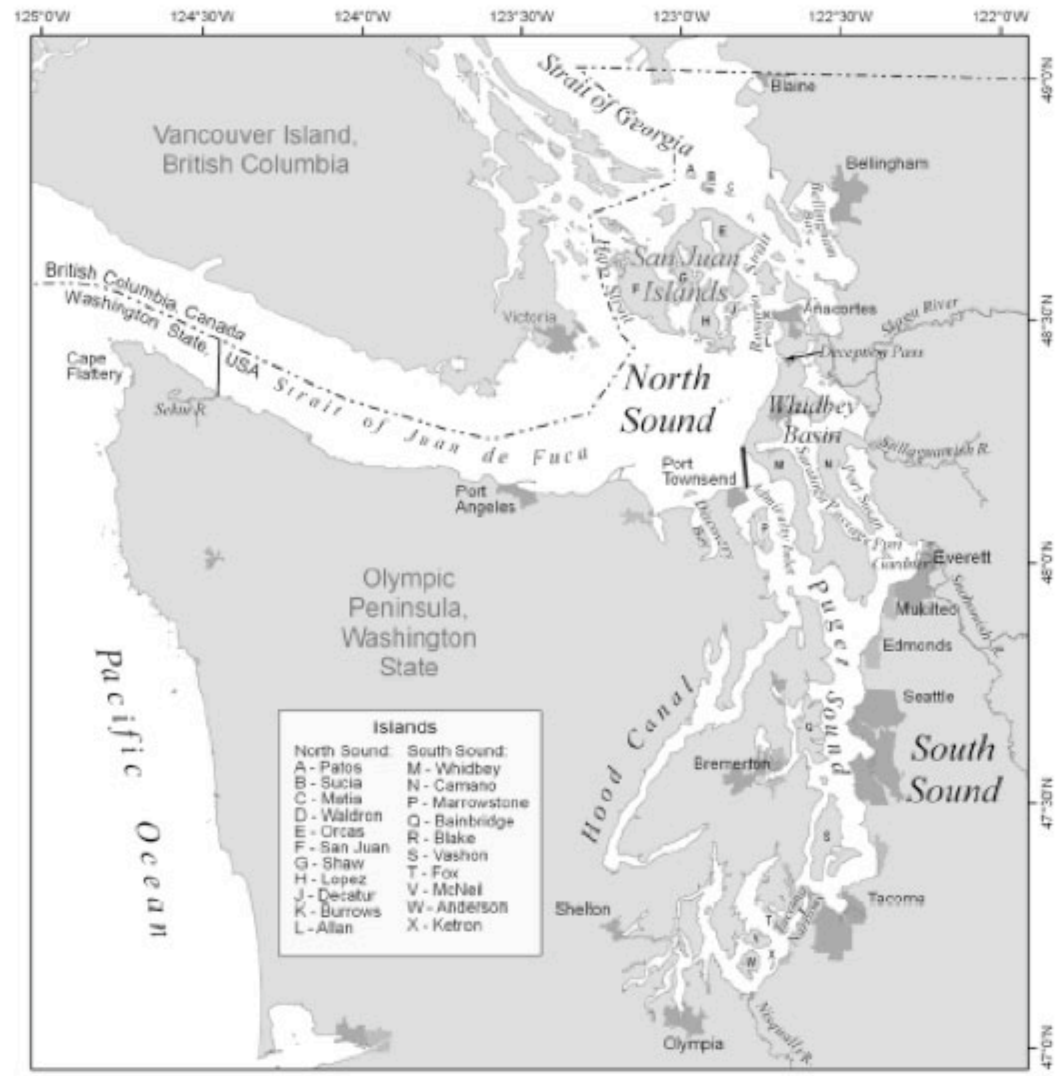


Figure 1. Map of Puget Sound showing North Sound and South Sound designations relevant to rockfish management (Reprinted from Palsson et al. 2009 with permission from .)

Currently, the Washington Department of Fish and Wildlife (WDFW) collects two types of data on rockfish in Puget Sound: those that are dependent upon information obtained from commercial and recreational fisheries (fishery-dependent data) and those that are based upon population surveys conducted by WDFW (fishery-independent data). The estimates of commercially removed biomass in Puget Sound are thought to be fairly accurate because documentation began in 1955 while recreational take has been monitored less consistently (Palsson et al. 2009). However, demographic data from the recreational fishery that inform the assessments of stock status for copper and quillback rockfish are collected by samplers trained by WDFW (Palsson et al. 2009). To obtain independent estimates of population abundances of rockfish, WDFW began conducting bottom trawls at a suite of sites in Puget Sound in 1987. The number of trawls for a given region has varied substantially over time (Palsson et al. 2009). Underwater video surveys are used to estimate biomass, density and distribution of rockfish. SCUBA transects are conducted along 30 m transects at approximately 25 sites in the North and South regions of Puget Sound (Palsson et al. 2009).

Using the abundances and trends from all available fishery-independent data, Palsson et al.(2009) classified each rockfish species as Healthy, Precautionary, Vulnerable or Depleted. These status categories are based on those used by the American Fisheries Society (Musick 1999). For the two rockfish species for which demographic data were most available (quillback and copper rockfish), designations were made based on current Spawners per Recruit (SPR) relative to 1970s SPR (proxy for an unfished population) and 1999 SPR (Palsson et al. 2009).

Status

The removal of rockfish from Puget Sound through recreational and commercial fisheries increased substantially after the Boldt Decision in 1974 when fishing restrictions were increased for salmon while rockfish were identified as a recommended alternative. Due to general declines in rockfish catches on the outer coast and to the petition for federal listing of 14 rockfish species found in the Puget Sound (Stout et al. 2001), commercial fishing for rockfish in Puget Sound has been restricted since 1999 and commercial catches have been negligible in recent years (Palsson et al. 2009). In 2002, any take of the yelloweye rockfish (*Sebastes ruberrimus*) and canary rockfish (*S. pinniger*) became prohibited. In 2004, the recreational daily limit on other rockfish species was reduced to a single fish (Palsson et al. 2009). In 2009, the Puget Sound populations of yelloweye and canary rockfish were federally listed as Threatened under the Endangered Species Act (ESA) and the bocaccio (*S. paucispinis*) was listed as Endangered. In addition to these federal listings, 14 of the 17 stocks of rockfishes in the North Puget Sound and 11 of the 15 stocks in the South Sound are currently designated by WDFW as Precautionary, Vulnerable or Depleted (Table 1).

Table 1. Summary of the status of rockfish stocks in Puget Sound (WDFW) (Palsson et al. 2009).

Species	North Sound	South Sound
Copper rockfish	Precautionary	Vulnerable
Quillback rockfish	Vulnerable	Depleted
Brown rockfish	Precautionary	Precautionary
Black rockfish	Precautionary	Precautionary
Yelloweye rockfish	Depleted	Depleted
Yellowtail rockfish	Precautionary	Precautionary
Canary rockfish	Depleted	Depleted
Bocaccio	Precautionary	Precautionary
Redstripe rockfish	Healthy	Healthy
Greenstriped rockfish	Healthy	Healthy
Splitnose rockfish	Precautionary	Precautionary
Shortspine thornyhead	Healthy	Healthy
Tiger rockfish	Precautionary	Precautionary
China rockfish	Precautionary	Not Present
Blue rockfish	Precautionary	Not Present
Vermilion rockfish	Precautionary	Precautionary
Puget Sound rockfish	Precautionary	Healthy
Number Healthy	3	4
Number Precautionary	11	7
Number Vulnerable	1	1
Number Depleted	2	3
Total Stocks Examined	17	15

A new management plan has recently been proposed by WDFW which outlines several possible management options for rockfish in Puget Sound and is currently under review (WDFW 2009). One of the key components of this plan is the recommendation that quillback, copper, black, yelloweye, bocaccio, canary and Puget Sound rockfish be managed as individual species due to their importance to recreational fisheries, conservation concerns, or ecological importance (WDFW 2009) (Table 2). In addition to these proposed changes in management, there are currently 16 marine reserves throughout Puget Sound that include the rocky habitat thought to be beneficial for rockfish.

Table 2. Proposed species of interest, habitats and reason for their selections in the Draft Puget Sound Rockfish Management Plant (WDFW).

SPECIES	COMPLEX	REASON
Copper rockfish	Nearshore	Important in recreational fisheries
Quillback rockfish	Nearshore	Important to recreational fisheries
Black rockfish	Pelagic	Important to recreational fisheries
Yelloweye rockfish	Deepwater	Conservation concerns, past economic importance
Bocaccio rockfish	Deepwater	Conservation concerns
Canary rockfish	Deepwater	Conservation concerns, past economic importance
Puget Sound rockfish (*Sebastes empheus)*	Nearshore	Important forage item

Trends

Both commercial and recreational catches of rockfishes have substantially declined since the mid 1980s and 1990s in both the North and South Puget Sound (Palsson et al. 2009) (Figure 2). Bottom trawl survey data also show declines in the harvested species of rockfishes; the two species that have increased over time (redstripe rockfish, *S. proriger* and Puget Sound rockfish, *S. emphaeus*) are smaller-bodied fish that are not harvested (Palsson et al. 2009)(Figure 3). The estimated SPR ratios for copper and quillback rockfish in the North and South Sound have also declined dramatically from 1970s to 1999 in both the North and South Sounds (Palsson et al. 2009)(Figure 4). This metric is important because it reflects the effect of fishing pressure on the reproductive capacity of a harvested population.

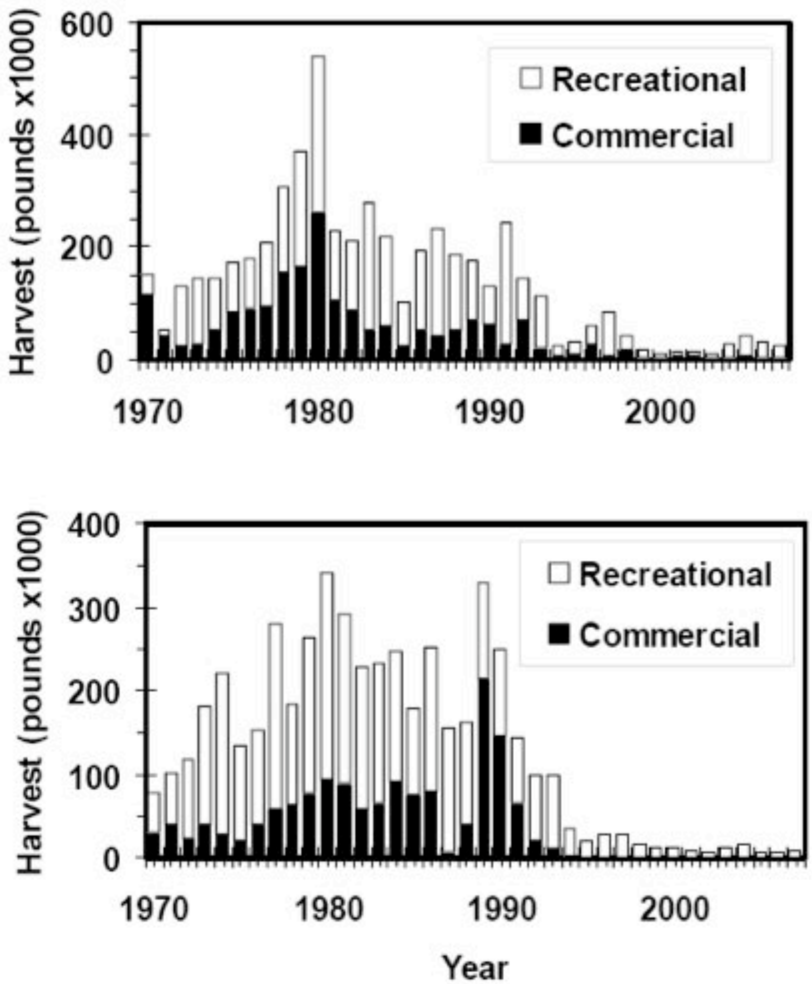


Figure 2. Total annual recreational (white) and commercial (black) harvest in pounds estimated by WDFW from North Puget Sound (top) and South Puget Sound (bottom) (Reprinted from Palsson et al. 2009 with permission from Washington Department of Fish and Wildlife.)

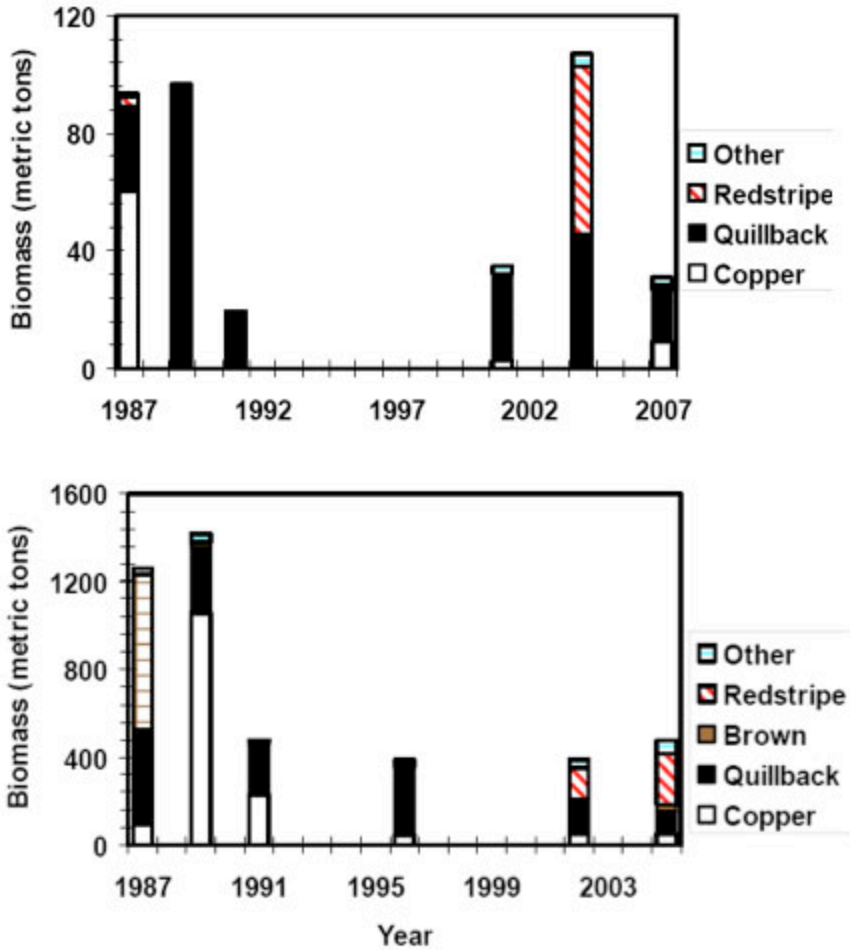


Figure 3. Biomass estimates (metric tons) from WDFW bottom trawl surveys from the Georgia Basin and East Juan de Fuca regions of North Sound (top) and South Sound (bottom) (Reprinted from Palsson et al. 2009 with permission from Washington Department of Fish and Wildlife.)

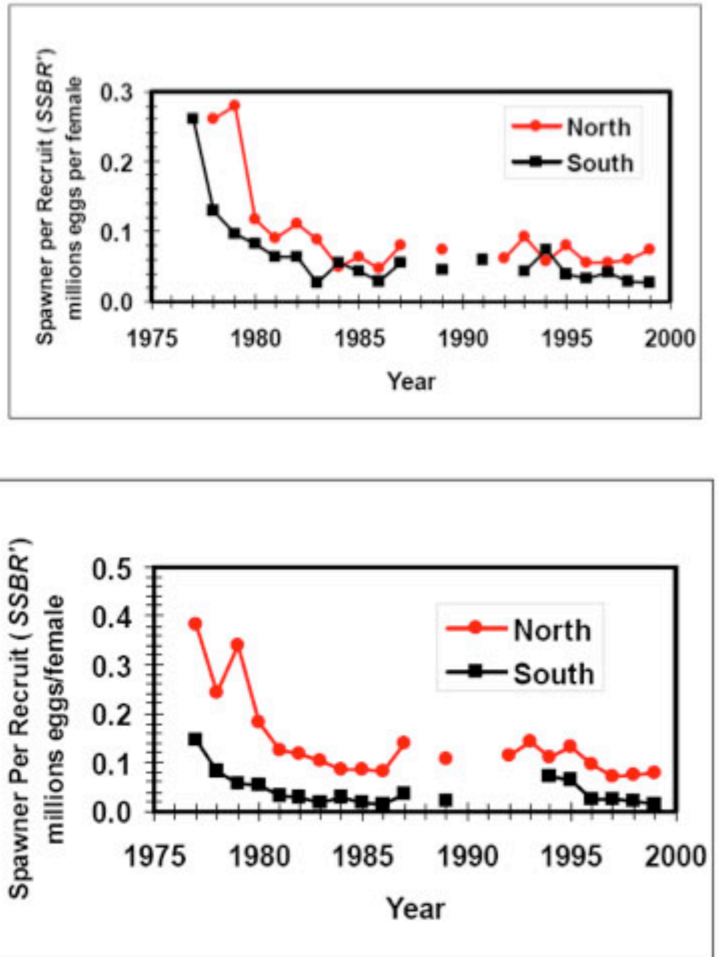


Figure 4. Spawner per Recruit Index (SSBR') for copper (top) and quillback (bottom) rockfishes in North Sound (red circles) and South Sound (black squares) (WDFW) (Reprinted from Palsson et al. 2009 with permission from Washington Department of Fish and Wildlife.)

Uncertainties

Many aspects of the ecology and biology of rockfish germane to their management in Puget Sound are not well understood. For example, ecological interactions such as predation may play important roles in determining the success of management strategies (e.g., Beaudreau and Essington 2007, Harvey et al. 2008), while demographic parameters such as age structure of populations (Berkeley et al. 2004, Berkeley 2006, Lucero 2009) or variability in the factors that drive recruitment rates are also likely to be quite important in driving the potential for rockfish recovery. Furthermore, while targeted exploitation of rockfishes in Puget Sound has diminished in recent years, the influence of continued threats such as pollution, altered food webs, incidental catch in recreational fisheries are not known.

Summary

Rockfish form a diverse assemblage of fish in Puget Sound and throughout their range. In Puget Sound, rockfish have abundances decreased substantially since quantitative monitoring began in the 1970s. These declines have resulted in the federal listing of three species under the Endangered Species Act. Because of their diversity in habitat use, ecology and life history, single-species approaches to rockfish management in Puget Sound are currently being considered.

Literature Cited

Beaudreau, A. H., and T. E. Essington. 2007. Spatial, temporal, and ontogenetic patterns of predation on rockfishes by lingcod. Transactions of the American Fisheries Society 136:1438-1452.

Berkeley, S. A. 2006. Pacific rockfish management: are we circling the wagons around the wrong paradigm? Bulletin of Marine Science 78:655-667.

Berkeley, S. A., C. Chapman, and S. M. Sogard. 2004. Maternal age as a determinant of larval growth and survival in a marine fish, *Sebastes melanops* Ecology 85:1258-1264.

Buonaccorsi, V. P., C. A. Kimbrell, E. A. Lynn, and R. D. Vetter. 2002. Population structure of copper rockfish (*Sebastes caurinus*) reflects postglacial colonization and contemporary patterns of larval dispersal. Canadian Journal of Fisheries & Aquatic Sciences 59:1375.

Burford, M. O. 2009. Demographic history, geographical distribution and reproductive isolation of distinct lineages of blue rockfish (*Sebastes mystinus*), a marine fish with a high dispersal potential. Journal of Evolutionary Biology 22:1471-1486.

Cope, J. M. 2004. Population genetics and phylogeography of the blue rockfish (*Sebastes mystinus*) from Washington to California. Canadian Journal of Fisheries & Aquatic Sciences 61:332-342.

Harvey, C. J., K. Gross, V. H. Simon, and J. Hastie. 2008. Trophic and fishery interactions between Pacific hake and rockfish: effect on rockfish population rebuilding times. Marine Ecology Progress Series 365:165-176.

Lance, M. M., and S. J. Jeffries. 2007. Temporal and spatial variability of harbor seal diet in the San Juan Island archipelago. Research Agreement No. K004431-25, Contract Report to SeaDoc Society, Olympia, WA.

Love, M. S., M. Yoklavich, and L. K. Thorsteinson. 2002. The rockfishes of the northeast Pacific. University of California Press, Berkeley, CA.

Lucero, Y. 2009. A multivariate stock-recruitment function for cohorts with sympatric subclasses: application to maternal effects in rockfish (genus *Sebastes*). Canadian Journal of Fisheries and Aquatic Sciences 66:557-564.

Miller, J. A., M. A. Banks, D. Gomez-Uchida, and A. L. Shanks. 2005. A comparison of population structure in black rockfish (*Sebastes melanops*) as determined with otolith microchemistry and microsatellite DNA. Canadian Journal of Fisheries and Aquatic Sciences 62:2189-2198.

Musick, J. A. 1999. Criteria to define extinction risk in marine fishes - The American Fisheries Society initiative. Fisheries 24:6-14.

Musick, J. A., M. M. Harbin, S. A. Berkeley, G. H. Burgess, A. M. Eklund, L. Findley, R. G. Gilmore, J. T. Golden, D. S. Ha, G. R. Huntsman, J. C. McGovern, G. R. Sedberry, S. J. Parker, S. G. Poss, E. Sala, T. W. Schmidt, H. Weeks, and S. G. Wright. 2000. Marine, Estuarine, and Diadromous Fish Stocks at Risk of Extinction in North America (Exclusive of Pacific Salmonids). Fisheries 25:6-30.

Palsson, W. A., T. S. Tsou, G. G. Bargmann, R. M. Buckley, J. E. West, M. L. Mills, Y. W. Cheng, and R. E. Pacunski. 2009. The biology and assessment of rockfishes in Puget Sound. Washington Department of Fish and Wildlife, Olympia, WA.

Parker, S. J., S. A. Berkeley, J. T. Golden, D. R. Gunderson, J. Heifetz, M. A. Hixon, R. Larson, B. M. Leaman, M. S. Love, J. A. Musick, V. M. O'Connell, S. Ralston, H. J. Weeks, and M. M. Yoklavich. 2000. Management of Pacific Rockfish. Fisheries 25:22-30.

Seeb, L. W. 1998. Gene flow and introgression within and among three species of rockfishes, *Sebastes auriculatus*, *S. caurinus*, and *S. maliger*. J Hered 89:393-403.

Stout, H. A., B. B. McCain, R. D. Vetter, T. L. Builder, W. H. Lenarz, L. L. Johnson, and R. D. Methot. 2001. Status review of Copper rockfish, Quillback rockfish and Brown rockfish in Puget Sound, Washington. NOAA Technical Memo, U.S. Department of Commerce.

WDFW. 2009. Draft Puget Sound Rockfish Conservation Plan. Washington Department of Fish and Wildlife, Olympia, WA.

Salmonids

Background

Fish in the family Salmonidae (salmon, trout, and charr) are unique in their cultural, economic and ecological role in Puget Sound. Because they utilize a very wide range of aquatic habitat types throughout their life history, they play potentially integral roles in the upland freshwater, nearshore and pelagic marine ecosystems and food webs of Puget Sound. They also provide key trophic links between habitats through their migratory behavior. While there is much variation in the behavior and ecology within and among the different salmonid species in Puget Sound, they typically use freshwater habitats to spawn, after which juveniles emerge and eventually migrate to nearshore estuaries or directly to marine pelagic habitats. Salmonids spend up to several years in marine habitats prior to returning to their natal watershed to spawn. Each life phase is thus dependent on a different suite of abiotic and biotic processes for survival. The use of nearshore habitats by juvenile salmon is thought to be a critical aspect of their capability to ultimately return and spawn (Fresh 2006). Available spawning habitat, appropriate water temperature and flow, and oceanic conditions are also important for salmonid survival and the degree of use of each type of habitat varies dramatically across the salmonid species.

The watersheds and nearshore habitats of Puget Sound currently support 8 species of salmon, trout, and charr (NOAA 2007)(Figure 1), four of which are listed as Threatened under the Endangered Species Act (ESA). These are Chinook salmon (*Oncorhynchus tshawytscha*), chum salmon (O. keta), bull trout (*Salvelinus confluentus*) and steelhead (*O. mykiss*). The recovery plan for Chinook, Hood Canal Summer Chum and bull trout put forth by Shared Strategy for Puget Sound and the Puget Sound Technical Recovery Team was adopted by NOAA Fisheries in 2007. The recovery strategy for these species is based upon the underlying principles of 1) abundance (the number of spawners); 2) productivity (the number of returning fish produced by each spawner); 3) spatial distribution (the geographic distribution of fish populations); and 4) diversity (of the genetic, physiological and morphological attributes)(NOAA 2007).

Data are collected on salmonid abundances in Puget Sound by a variety of local, state and federal agencies including Washington Department of Fish and Wildlife(WDFW), NOAA Fisheries and the US Fish and Wildlife Service. Spawner abundances are typically estimated in the field by counting the number of nests (redds) or by counting the number of spawning and/or dead fish. WDFW maintains an online database of watershed-specific spawner abundances (Salmonscape) and also conducts stock status estimates (Salmonid Stock Inventory) whereby each spawning stock is designated as Healthy, Depressed, Critical, Extinct or Unknown based upon recent abundance trends for all species except for Chinook salmon (WDFW 2002). The most recent stock inventory categorization utilized trend data from the mid 1980s to 2000 or 2001 (WDFW 2002).

Chinook salmon

The largest of the salmonids, Chinook salmon typically spawn in larger rivers and their tributaries, utilizing deeper water and larger gravel for egg burial than their congeners. While Chinook fry are often classified as either ocean-type or stream-type depending on the timing of their initial downstream migration, in Puget Sound this has further been subdivided into four broad types of strategies based upon general timing emigration from both freshwater and estuarine habitats prior to eventually migrating to coastal oceanic waters (Fresh 2006). These range from up to a year spent in natal freshwater streams with very little time spent migrating through estuarine habitat to very early emigration from freshwater with up to 120 days spent rearing in natal estuaries and tidal wetlands (summarized in Fresh 2006). This diversity is thought to be critical for the continued survival of this species (NOAA 2007). There is emerging evidence that some Chinook salmon remain in Puget Sound waters as residents with little or no time spent in oceanic waters (O'Neill and West 2009). Following entry into the open ocean via the Strait of Juan de Fuca, Chinook salmon are believed to migrate mostly northwards towards British Columbia and Alaska, remaining on the continental shelf where they typically spend 2-4 years before returning to their natal stream to spawn and die (Quinn 2005, Quinn et al. 2005).

Hood Canal Summer Chum salmon

Chum salmon typically spawn in the lower reaches of rivers with fry leaving fresh water for estuarine habitats within days of emergence. In Puget Sound, they can either remain in their natal estuaries or transition to other estuaries and nearshore habitats to rear before entering oceanic waters. While utilizing estuary habitats, chum salmon primarily feed upon epibenthic invertebrates associated with eelgrass (summarized in Fresh 2006).

Steelhead

Unlike Chinook and chum salmon, steelhead are iteroparous, displaying a diverse suite of life history variations with variable time spent in fresh, salt water and estuarine environments. They are thought to leave coastal waters immediately after entering the ocean, occupying marine habitats distinct from that of their congeners, spending 1-3 years at sea (Quinn et al. 2005, Hard et al. 2007). While little is known about the oceanic migration patterns of Puget Sound steelhead, there is evidence that they travel to the Central North Pacific (reviewed in Hard et al. 2007). The resident (non-migratory) form of steelhead (rainbow trout) is also present in Puget Sound (Hard et al. 2007).

Bull trout

Like steelhead, Bull trout are iteroparous and long lived, potentially spawning in their natal streams several times throughout their lifetime. Like cutthroat trout, bull trout often occupy nearshore marine habitats during their short seaward migration.

Status

Chinook salmon

Listed as Threatened in 1999, Chinook salmon currently maintain 22 of the estimated 30-37 historically present spawning populations that utilize rivers and streams throughout Puget Sound. (NOAA 2007)(Figure 1, Table 1). Many of the populations lost were those that spawned earlier in the spawning season when water levels are typically lower and temperatures are higher (NOAA 2007). There is also evidence that the life history variants that spend the greatest time in freshwater (stream-type) have been severely reduced in recent years, likely because these variants are particularly susceptible to watershed alteration such as damming (Beechie et al. 2006). Furthermore, the few remaining locations where the stream-type life history has persisted appear to be largely dependent on snowmelt for their water and thus may be particularly susceptible to the effects of global climate change (Beechie et al. 2006). The Puget Sound Technical Recovery Team (PSTRT) estimated the historic spawner abundances by modeling the number of individuals each watershed could support based upon habitat characteristics (NOAA 2007). For all populations for which this analysis has been conducted, current population levels are substantially less than the estimated historic levels (NOAA 2007)(Figure 2).

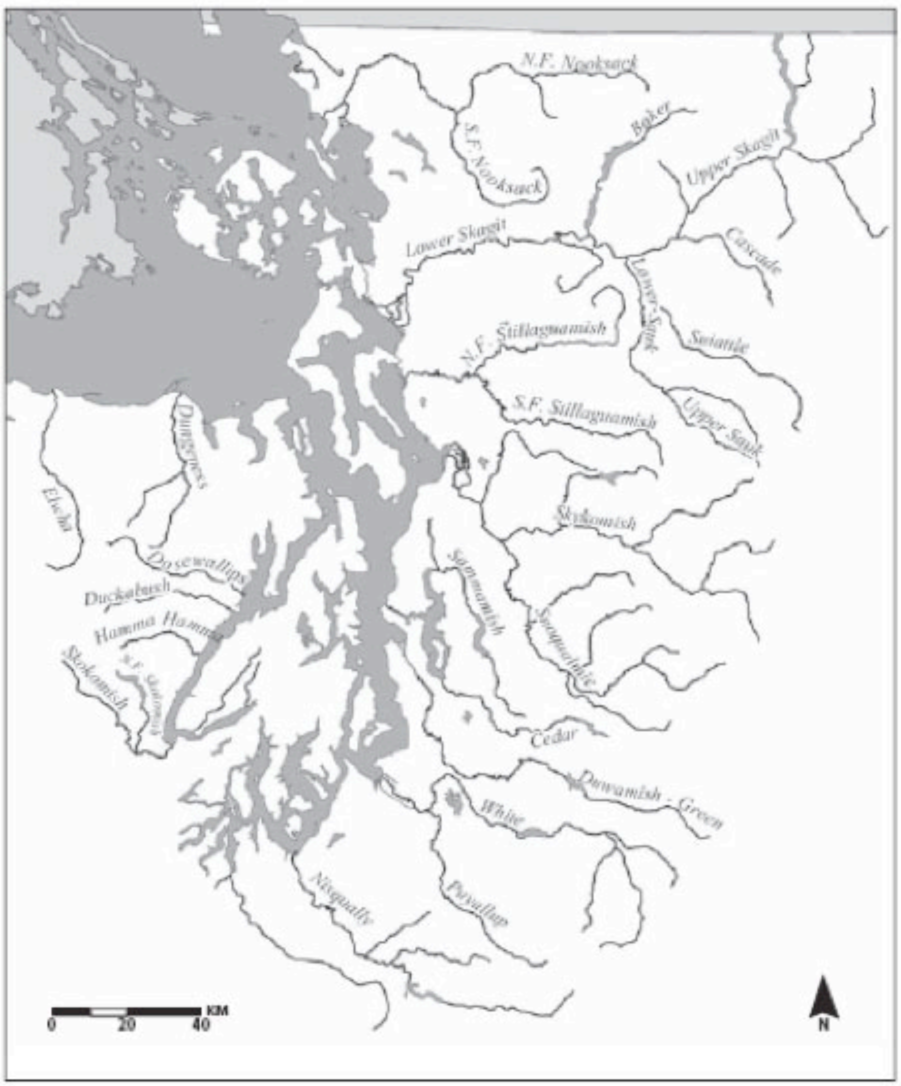


Figure 1. Map of major watersheds in Puget Sound utilized by salmonids (Reprinted from NOAA 2007; courtesy of NOAA Fisheries).

Table 1. Extant populations of Chinook salmon in Puget Sound (NOAA Salmon Recovery Plan 2007).

Geographic Region	Populations Remaining
Strait of Georgia This area includes the Nooksack River and the San Juan Islands. It is an area greatly influenced by the Fraser River and is utilized extensively for forage and migration by many Puget Sound populations.	North Fork Nooksack South Fork Nooksack
Strait of Juan de Fuca This region includes the rivers draining the north slopes of the Olympic mountains, and draining into the eastern Strait of Juan de Fuca. Nearshore areas along the Strait are considered to be a major migratory corridor.	Elwha Dungeness
Hood Canal The east face of the Olympic mountain range and small streams along the western Kitsap Peninsula drain into this distinct estuary.	Skokomish Mid Hood Canal (incl. Dosewallips, Duckabush and Hamma Hamma)
Whidbey Basin The Whidbey basin is the main estuarine area for the major Chinook-producing rivers in Puget Sound, and the migratory crossroads for most Puget Sound populations.	Skykomish Snoqualmie North and South Fork Stillaguamish Upper and Lower Skagit Upper and Lower Sauk Suiattle Cascade
Central/South Basin These basins were combined into a single geographic unit largely to reflect correlated risks from volcanic activity and urban-related effects.	Cedar River North Lake Washington Green/Duwamish Puyallup White Nisqually

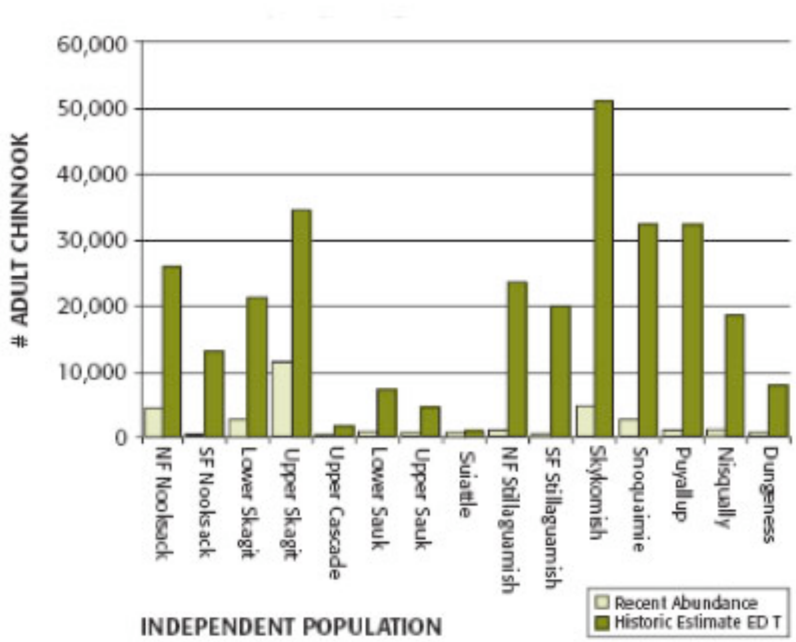


Figure 2. Comparison of recent (2000-2004) geometric mean of naturally spawning Puget Sound Chinook populations to estimates of historic capacity of in some Puget Sound watersheds using Ecosystem Diagnostic and Treatment (EDT) habitat models (Reprinted from NOAA Salmon 2007; courtesy of NOAA Fisheries).

Hood Canal Summer Chum

The summer run of Hood Canal chum salmon was listed as Threatened in 1999. A primary factor in this designation was the recognition that this stock comprises an important and distinct life history strategy within the species (NOAA 2007). Of the 16 historic spawning stocks of Hood Canal summer chum, eight are extant (NOAA 2007)(Table 2). In a recent review of this Threatened Evolutionarily Significant Unit (ESU), two genetically distinct populations were identified: a Strait of Juan de Fuca population (which includes the extant spawning aggregations Jimmycomelately, Snow, Salmon and Chimacum creeks) and a Hood Canal population (which includes the extant spawning aggregations Big and Little Quilcene, Dosewallips, Duckabush, Hamma Hamma, Union and Lilliwaup watershds)(Sands et al. 2009)(Figure 3). Maintaining diversity within and between these newly two newly identified populations will now be incorporated into the recovery goals for Hood Canal Summer Chum (Sands et al. 2009).

Table 2. Current (extant) and extinct populations of Hood Canal summer chum and supplementation/reintroduction programs (NOAA Salmon Recovery Plan 2007).

Population	Status	Supplementation/Reintroduction Program
Union River	Extant	Supplementation program began in 2000
Lilliwaup Creek	Extant	Supplementation program began in 1992
Hamma Hamma River	Extant	Supplementation program began in 1997
Duckabush River	Extant	—
Dosewallips River	Extant	—
Big/Little Quilcene River	Extant	Supplementation program began in 1992
Snow/Salmon Creeks	Extant	Supp. program began in 1992 in Salmon
Jimmycomelately Creek	Extant	Supplementation program began in 1999
Dungeness River	Unknown	—
Big Beef Creek	Extinct	Reintroduction program began in 1996
Anderson Creek	Extinct	—
Dewatto Creek	Extinct	Natural re-colonization occurring
Tahuya River	Extinct	—
Skokomish River	Extinct	—
Finch Creek	Extinct	—
Chimacum Creek	Extinct	Reintroduction program

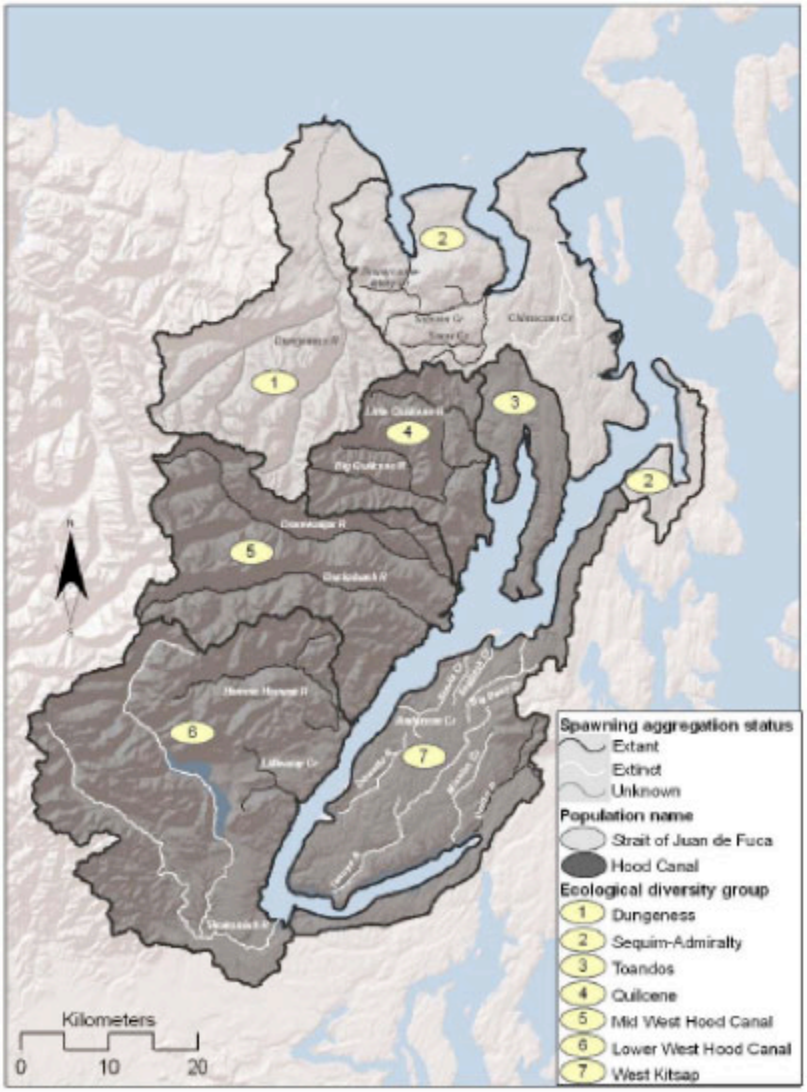


Figure 3. The two populations of the Hood Canal Summer Chum salmon ESU, including streams with spawning aggregations and seven ecological diversity groups (Reprinted from Sands et al. 2009; courtesy of NOAA Fisheries).

Steelhead

Steelhead in the Puget Sound ESU were federally listed as Threatened in 2007 (Hard et al. 2007). WDFW currently lists 53 spawning populations of steelhead in Puget Sound, the majority of which return in the winter to spawn. Less well studied and less abundant, the remaining populations return in the summer and are typically found in the northern portions of Puget Sound (Hard et al. 2007). The two largest populations of winter steelhead are also in the northern part of the sound, in the Skagit and Snohomish rivers (Hard et al. 2007)(Table 3).

Table 3. Geometric mean estimates of escapements of Puget Sound populations for all years of data (from ca. 1980 – 2004 for most populations) and for the 5 most recent years (2000 – 2004).

Estimates are based on hatchery and natural spawner (H+N columns) or only on natural spawners (N columns). Hatchery fish are not included in the Puget Sound ESU. NPS = Northern Puget Sound, SPS = Southern Puget Sound, HC = Hood Canal, SJF = Strait of Juan de Fuca, SSH = summer run steelhead, WSH = winter run steelhead, N/A = data not available (Hard et al. 2007).

Region	Run type	Population	H+N, all years	H+N, 5 years	N, all years	N, 5 years
NPS	SSH	Canyon	N/A	N/A	N/A	N/A
NPS	SSH	Skagit	N/A	N/A	N/A	N/A
NPS	SSH	Snohomish	N/A	N/A	N/A	N/A
NPS	SSH	Stillaguamish	N/A	N/A	N/A	N/A
NPS	WSH	Canyon	N/A	N/A	N/A	N/A
NPS	WSH	Dakota	N/A	N/A	N/A	N/A
NPS	WSH	Nooksack	N/A	N/A	N/A	N/A
NPS	WSH	Samish	684.2	852.2	500.8	852.2
NPS	WSH	Skagit	7,720.4	5,608.5	6,993.9	5,418.8
NPS	WSH	Snohomish	5,283.0	3,230.1	5,283.0	3,230.1
NPS	WSH	Stillaguamish	1,027.7	550.2	1,027.7	550.2
NPS	SSH	Tolt	129.2	119.0	129.2	119.0
SPS	SSH	Green	N/A	N/A	N/A	N/A
SPS	WSH	Cedar	137.9	36.8	137.9	36.8
SPS	WSH	Green	2,050.6	1,625.5	1,802.1	1,619.7
SPS	WSH	Lk. Washington	247.1	36.8	308.0	36.8
SPS	WSH	Nisqually	1,136.7	392.4	1,115.9	392.4
SPS	WSH	Puyallup	1,881.5	1,001.0	1,714.4	907.3
HC	WSH	Dewatto	27.0	24.7	24.0	24.7
HC	WSH	Dosewallips	70.6	76.7	70.6	76.7
HC	WSH	Duckabush	16.6	17.7	16.6	17.7
HC	WSH	Hamma Hamma	29.6	51.9	29.6	51.9
HC	WSH	Quilcene	16.8	15.1	16.8	15.1
HC	WSH	Skokomish	439.3	202.8	439.3	202.8
HC	WSH	Tahuya	131.8	117.0	113.9	117.0
HC	WSH	Union	57.1	55.3	55.0	55.3
SJF	SSH	Elwha	N/A	N/A	N/A	N/A
SJF	WSH	Dungeness	311.2	173.8	311.2	173.8
SJF	WSH	Elwha	459.5	210.0	N/A	N/A
SJF	WSH	McDonald	N/A	N/A	149.8	96.1
SJF	WSH	Morse	132.6	103.0	105.8	103.0

Bull trout

Bull trout in Washington, including the Puget Sound Distinct Population Segment (DPS), were also listed as Threatened in 1999. The US Fish and Wildlife Service conducted an analysis of vulnerability to stochastic events across the spawning stocks of bull trout in Puget Sound, finding the Snohomish/Skyhomish, the Stillaguamish, and the Chester Morse Lake spawning stocks to be at the greatest risk (NOAA 2007)(Table 4).

Table 4. Bull trout risk levels for watersheds in Puget Sound (USFWS data)(NOAA Salmon Recovery Plan 2007)

Core Areas	Local and Potential Local Populations	Information on Abundance, Trends and Distribution	Risk from Stochastic Events
Chilliwack	Little Chilliwack River Upper Chilliwack River Selesia Creek (British Columbia & US) Depot Creek (BC & US) Airplane Creek (BC) Borden Creek (BC) Centre Creek (BC) Foley Creek (BC) Nesakwatch Creek (BC) Paleface Creek (BC)	Chilliwack Lake is an important source of rearing and forage for most local populations.	Intermediate risk if only the US populations are considered. Diminished risk if both US and Canadian populations are considered.
Nooksack	Lower Canyon Creek Glacier Creek Lower Middle Fork Nooksack R Upper MF Nooksack River Lower North Fork Nooksack R Middle NF Nooksack River Upper NF Nooksack River Upper South Fork Nooksack R Lower SF Nooksack River Wanlick Creek	Spawning occurs in all three forks of the Nooksack River and its tributaries. Fewer than 1000 spawners; most local populations have less than 100 adults.	Intermediate Risk
Lower Skagit	Bacon Creek Baker Lake Buck Creek Cascade River South Fork Cascade River Downey Creek Goodell Creek Illabot Creek Lime Creek Milk Creek Newhalem Creek Forks of Sauk River Upper South Fork Sauk River Straight Creek Upper Sulattle River Sulphus Creek Tenas Creek Lower White Chuck River Upper White Chuck River Sulphur Creek -Lake Shannon (potential local population) Stetattle Creek-Gorge Lake (potential local population)	Bull trout are known to spawn and rear in at least 19 streams/ stream complexes. This core area supports a spawning population of migrating bull trout numbering in the thousands. Connectivity and diversity of habitats are excellent except portions modified by dams. High abundance of pink salmon for forage.	Diminished Risk
Upper Skagit	Big Beaver Creek Little Beaver Creek Lightning Creek Panther Creek Pierce Creek Ruby Creek Silver Creek Thunder Creek (Diablo Lake) Deer Creek (Diablo Lake) (potential local population) Skagit River (BC) East Fork Skagit River (BC) Klesilkwa River (BC) Nepopekum Creek (BC) Skaist River (BC) Sumallo River (BC)	Populations are well distributed. British Columbia portion presumed healthy; status is generally unknown. 2 areas of concern due to lack of connectivity: Diablo Lake and Gorge Lake.	Intermediate risk if only the US populations are considered. Diminished risk if both US and Canadian populations are considered.
Stillaguamish	Upper Deer Creek South Fork Canyon Creek North Fork Stillaguamish River South Fork Stillaguamish River	Few known spawning areas. Fewer than 1000 spawners; most local populations have less than 100 adults. Snorkel surveys have found greater than 100 adults in the North Fork Stillaguamish R.	Increased risk
Snohomish-Skykomish	South Fork Skykomish River Salmon Creek Troublesome Creek (primarily a resident population)	number of adult spawners is 500-1000. System has no lakes. Large portion of migratory segment are anadromous. North Fork Sky considered healthy by WDFW with 470-650 individuals on average, based on redd counts. South Fork Sky considered healthy by WDFW due to increasing numbers, and recolonization is occurring.	Increased risk
Chester Morse Lake	Boulder Creek Upper Cedar River Rex River Rack Creek Shotgun Creek (potential local population)	Area has few known spawning areas. Surveys in 2000-2002 documented 236-504 redds, with estimated 500-1000 spawners. Upper Cedar River and Rex River are the primary local populations in this core area. Upper Cedar River is the only known self-sustaining population in the Lake WA basin.	Increased risk
Puyallup	Carbon River Greenwater River Upper Puyallup and Mowich Rivers Upper White River West Fork White River Clearwater River (potential local population)	Fewer than 1000 spawners; most local populations have less than 100 adults. Known spawning areas are few and not widespread. Area has a low number of local populations. Portions within the National Park and wilderness area provide pristine habitat.	Intermediate risk

Trends

Chinook salmon

An analysis of 5-year population growth trends for Chinook salmon from 1986 - 2004 was conducted by NOAA fisheries. Of those populations that had been declining from 1986 – 1990, many exhibited positive growth over 1994 – 1998. (NOAA 2007) (Table 5). While productivity was not calculated for the most recent time period (2000-2004), the population means for this period were, in many cases, higher than that observed previously (NOAA 2007)(Table 5). Despite this positive trend, many populations remained low, including the Dungeness River and Skokomish spawning stocks (NOAA 2007)(Table 5).

Table 5. Geometric mean (5 year periods) of spawning abundances, hatchery contribution and productivity (number of return spawners per parent spawner) in Puget Sound Chinook Populations (NOAA Salmon Recovery Plan 2007).

	1986-1990			1994-1998			2000-2004	
Populations	Geometric Mean	% Hatchery Contribution	Productivity	Geometric Mean	% Hatchery Contribution	Productivity	Geometric Mean	% Hatchery Contribution
North + Middle Fork Nooksack	140	21%	1.29	263	67%	0.45	4,232	94%
South Fork Nooksack	243	7%	0.60	181	35%	1.20	303	46%
Lower Skagit	2,732	1%	0.59	974	1%	3.15	2,597	2%
Upper Skagit	8,020	2%	0.69	6,388	1%	1.60	12,116	4%
Upper Cascade	226	0%	0.88	241	0%	1.34	355	1%
Lower Sauk	888	0%	0.61	330	0%	2.35	825	0%
Upper Sauk	720	0%	0.57	245	0%	1.35	413	0%
Suiattle	687	0%	0.40	365	0%	1.20	409	0%
North Fork Stillaguamish	699	0%	0.92	862	35%	0.94	1,176	31%
South Fork Stillaguamish	257	0%	1.31	246	0%	1.22	205	0%
Skykomish	3,204	14%	0.52	3,172	52%	0.82	4,759	39%
Snoqualmie	907	12%	1.23	1,012	33%	1.68	2,446	14%
Sammamish	388	41%	0.28	145	74%	2.72	243	69%
Cedar	733	9%	0.51	391	17%	0.97	412	21%
Green/Duwamish	7,966	62%	0.50	7,060	71%	1.00	13,172	34%
White	73	56%	7.51	452	82%	1.49	1,417	28%
Puyallup	1,509	15%	1.86	1,657	40%	0.67	1,353	31%
Nisqually	602	3%	4.22	753	21%	1.38	1,295	25%
Skokomish	1,630	69%	0.48	866	69%	0.34	1,479	80%
Mid Hood Canal	87	26%	1.41	182	26%	1.31	202	46%
Dungeness	185	83%	0.12	101	83%	0.70	532	83%
Elwha Nat Spawners	2,055	34%	0.46	512	61%	1.03	847	54%
Elwha Nat+Hat Spawners	3,887	34%	0.67	1,679	61%	1.27	2,384	54%

Hood Canal Chum salmon

Population growth rates for Hood Canal summer chum salmon were all positive over short- time frames (1999-2002), but only two of the eight spawning aggregations (Union River and Big/Little Quilcene River) displayed positive growth rates over longer time scales (1970s – 2002) (Table 6). The latter two are both constituents of the Hood Canal genetically independent population (Sands et al. 2009), and experienced declines in the 1980- 1990s followed by recent increases (Sands et al. 2009)(Figure 4).

Table 6. Mean abundance of Hood Canal summer chum in each watershed and long-term (1970s – 2002) and short-term (1999 - 2002) population growth trends (NOAA Salmon Recovery Plan 2007).

Population	Geometric mean escapement (1999-2002)	Long Term Trend (a value of 1.0 indicates that the population is replacing itself)	Short Term Trend
Union River	594	1.08	1.10
Lilliwaup Creek	13	0.88	1.00*
Hamma Hamma River	558	0.90	1.20
Duckabush River	382	0.91	1.14
Dosewallips River	919	0.96	1.25
Big/Little Quilcene River	4,512	1.05	1.62
Snow/Salmon Creeks	1,521	0.99	1.24
Jimmycomelately Creek	10	0.88	0.82*
* Supplementation programs at Jimmycomelately and Lilliwaup reduced the number of spawners released to achieve escapement naturally.			

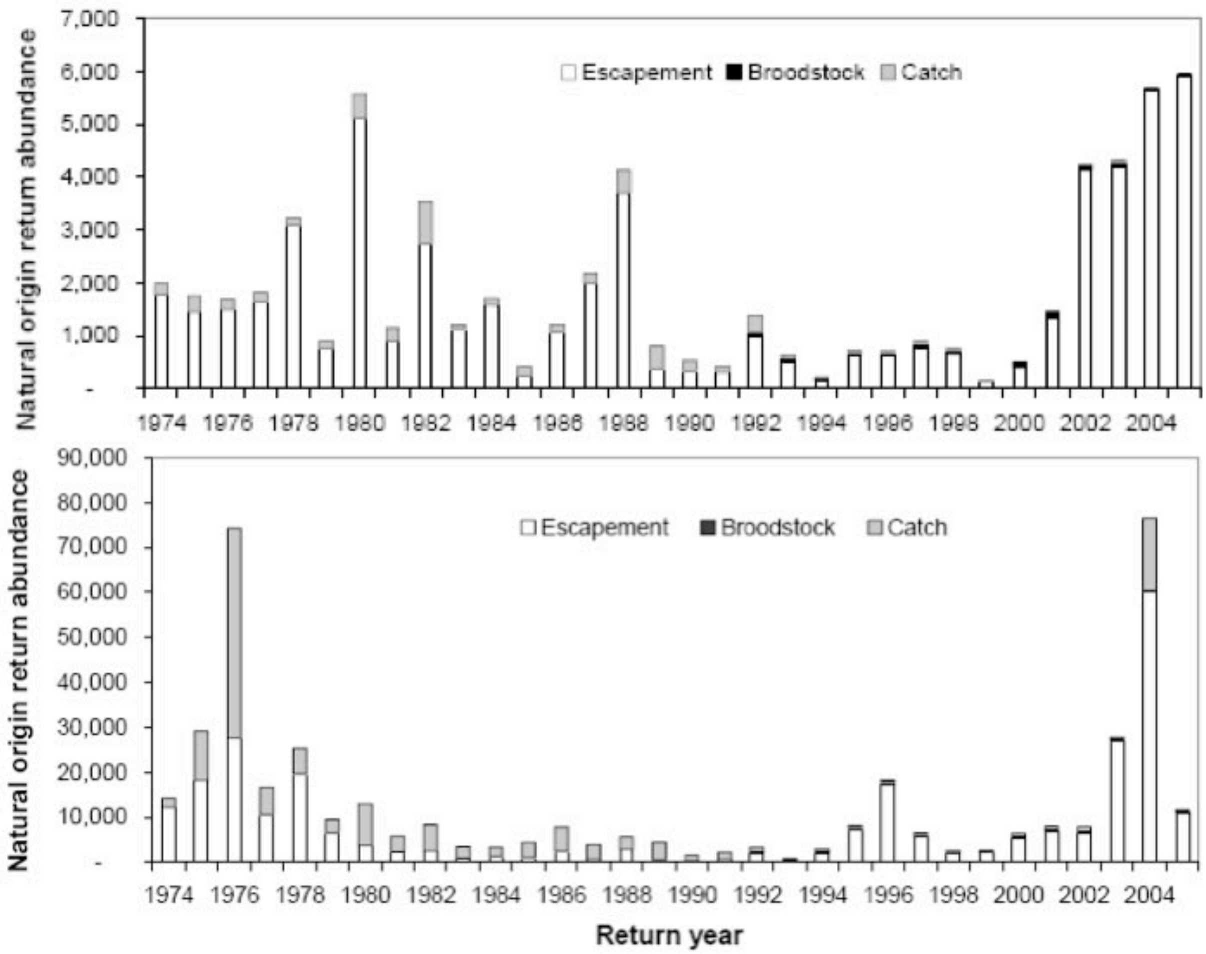


Figure 4. Annual return abundances of natural-origin summer chum salmon of the Strait of Juan de Fuca region (TOP) and the Hood Canal region (BOTTOM) from 1974 – 2005 (Reprinted from Sands et al. 2009; courtesy of NOAA Fisheries).

Steelhead Analyses utilizing all years of available data (ca. 1980 – 2004) and the 10 most recent years (1995-2004) indicated that most Puget Sound steelhead populations exhibited significantly declining trends in natural escapements, particularly in the southern Puget Sound (e.g., the Cedar, Lake Washington, Nisqually and Puyallup winter run populations) (Hard et al. 2007)(Table 7). Increasing populations were observed in the Samish and Hamma Hamma winter run populations (Hard et al. 2007)(Table 7).

Table 7. Estimates of temporal trends in escapement (E) and total run size(R) (log-transformed) for naturally produced Puget Sound. Positive values indicate a growing population, negative values indicate a declining one. Asterices indicate level of significance (Hard et al. 2007).

Region	Run type	Population	E, all years	E, 10 years	R, all years	R, 10 years
NPS[a]	SSH[b]	Canyon	N/A[c]	N/A	N/A	N/A
NPS	SSH	Skagit	N/A	N/A	N/A	N/A
NPS	SSH	Snohomish	N/A	N/A	N/A	N/A
NPS	SSH	Stillaguamish	N/A	N/A	N/A	N/A
NPS	WSH[d]	Canyon	N/A	N/A	N/A	N/A
NPS	WSH	Dakota	N/A	N/A	N/A	N/A
NPS	WSH	Nooksack	N/A	N/A	N/A	N/A
NPS	WSH	Samish	+0.067**	+0.061**	+0.019	+0.014
NPS	WSH	Skagit	−0.002	−0.010	−0.021	−0.056
NPS	WSH	Snohomish	−0.019	+0.035*	−0.086	N/A
NPS	WSH	Stillaguamish	−0.065****	N/A	−0.110*	N/A
NPS	SSH	Tolt	+0.025	+0.034	−0.107	−0.021
SPS[e]	SSH	Green	N/A	N/A	N/A	N/A
SPS	WSH	Cedar	−0.179**	N/A	−0.299*	N/A
SPS	WSH	Green	+0.008	−0.016**	−0.048	−0.069*
SPS	WSH	Lk. Washington	−0.180****	−0.215****	−0.300*	−0.274
SPS	WSH	Nisqually	−0.084****	−0.147****	−0.097	−0.159**
SPS	WSH	Puyallup	−0.062****	−0.074****	−0.103**	−0.103**
HC[f]	WSH	Dewatto	N/A	N/A	N/A	N/A
HC	WSH	Dosewallips	N/A	N/A	N/A	N/A
HC	WSH	Duckabush	+0.017	−0.018	+0.017	−0.019
HC	WSH	Hamma Hamma	+0.291*	+0.264	+0.291*	+0.264
HC	WSH	Quilcene	−0.006	N/A	−0.006	N/A
HC	WSH	Skokomish	−0.075****	−0.136**	−0.109*	−0.136**
HC	WSH	Tahuya	+0.009	−0.002	+0.004	−0.021
HC	WSH	Union	+0.008	+0.002	+0.008	+0.002
SJF[g]	SSH	Elwha	N/A	N/A	N/A	N/A
SJF	WSH	Dungeness	−0.076****	−0.093**	−0.083	−0.093
SJF	WSH	Elwha	N/A	N/A	N/A	N/A
SJF	WSH	McDonald	−0.031	+0.009	−0.362**	−0.221*
SJF	WSH	Morse	−0.006	−0.015	−0.030	−0.050

[a] NPS = Northern Puget Sound.
[b] SSH = Summer run steelhead.
[c] N/A = Data not available.
[d] WSH = Winter run steelhead.
[e] SPS = Southern Puget Sound.
[f] HC = Hood Canal.
[g] SJF = Strait of Juan de Fuca.

Bull trout

There is a paucity of reported data on the population trends of bull trout in Puget Sound.

Uncertainties

Because of the wide array of life history types exhibited and habitats utilized by salmonids, the list of human threats as well as environmental and ecological drivers of salmonid abundance is long. These include hydropower, harvest, reduction in freshwater habitat quality and quantity, interactions with other fish, birds and marine mammals, ocean conditions and negative impacts of hatchery-reared salmon (Ruckelshaus et al. 2002). These drivers likely apply to both listed and non-listed salmonids in Puget Sound.

Summary

Salmon and trout are key ecological, cultural and economic components of the Puget Sound ecosystem. The number of Chinook salmon has increased since being listed in 1999, although population numbers remain well below target abundances. Hood Canal Summer chum salmon populations have shown some increases since their listing. Population abundance data for the two listed trout and charr species have not been published in citable reports or other publications.

Literature Cited

Beechie, T., E. Buhle, M. Ruckelshaus, A. Fullerton, and L. Holsinger. 2006. Hydrologic regime and the conservation of salmon life history diversity. Biological Conservation 130:560-572.

Fresh, K. L. 2006. Juvenile Pacific salmon and the nearshore ecosystem of Puget Sound., Published by Seattle Districut, U.S Army Corps. of Engineeers, Seattle, WA.

Hard, J. J., J. M. Myers, M. J. Ford, R. G. Cope, G. R. Pess, R. S. Waples, G. A. Winans, B. A. Berejikian, F. W. Waknitz, P. B. Adams, P. A. Bisson, D. E. Campton, and R. R. Reisenbichler. 2007. Status review of Puget Sound steelhead (*Oncorhynchus mykiss*). U.S. Department of Commerce, NOAA Tech Memo, Seattle, WA.

NOAA. 2007. Puget Sound Salmon Recovery Plan. Shared Strategy for Puget Sound and NOAA Fisheries, Seattle, WA.

O'Neill, S. M., and J. E. West. 2009. Marine distribution, life history traits, and the accumulation of polychlorinated biphenyls in chinook salmon from Puget Sound, Washington. Transactions of the American Fisheries Society 138:616-632.

Quinn, T. P. 2005. The behavior and ecology of Pacific salmon and trout. American Fisheries Society, Bethesda, Md.

Quinn, T. P., B. R. Dickerson, and L. A. Vøllestad. 2005. Marine survival and distribution patterns of two Puget Sound hatchery populations of coho (*Oncorhynchus kisutch*) and chinook (*Oncorhynchus tshawytscha*) salmon. Fisheries Research 76:209-220.

Ruckelshaus, M. H., P. Levin, J. B. Johnson, and P. M. Kareiva. 2002. The Pacific salmon wars: What science brings to the challenge of recovering species. Annual Review of Ecology and Systematics 33:665-706.

Sands, N. J., K. Rawson, K. P. Currens, W. H. Graeber, M. H. Ruckelshaus, R. R. Fuerstenberg, and J. B. Scott. 2009. Determination of independent populations and viability criteria for the Hood Canal summer chum salmon evolutionarily significant unit. U.S. Department of Commerce, NOAA Fisheries, Seattle, WA.

WDFW. 2002. 2002 SaSI Introduction. Washington Department of Fish and Wildlife, Olympia, WA.

Marine birds

Background

Puget Sound is important for nesting, wintering, and migration of numerous bird species associated with the marine environment. More than 70 bird species regularly utilize Puget Sound during some or all stages of their life histories (Buchanan 2006), but only a portion of these are actively being investigated. Studies have focused primarily on abundance and distribution, habitat utilization, foraging behavior, and contamination levels.

Multispecies comparisons

Information pertaining to marine bird distribution and abundance prior to the 1970s resides primarily in anecdotal accounts (Rathbun 1915, Jewett 1953) and systematic surveys held during Christmas Bird Counts (CBCs), which became consistent and widespread in the 1960s. Since the 1970s, the most comprehensive census of marine birds in northern Puget Sound was conducted as part of the Marine Ecosystems Analysis (MESA) program of 1978-1979 (Wahl 1981). The MESA study was a large-scale survey jointly funded by the Department of Commerce (DOC) and the Environmental Protection Agency (EPA) as a response to oil spill threats in the Strait of Juan de Fuca. It included aerial, land-based, and ferry-based transect surveys north of Admiralty Inlet, within portions of the Straits of Juan de Fuca and Georgia, and the Canadian Gulf Islands. Notably, the study included only the southernmost portion of the Strait of Georgia and not Puget Sound itself.

Beginning in 1992, the Puget Sound Ambient Monitoring Program (PSAMP) began collecting observations of marine birds in the non-breeding season; this currently is the only source of continuous multi-species monitoring of marine birds in Puget Sound. The annual surveys consist of aerial transects covering nearshore habitat and offshore habitat/open waters throughout Puget Sound and the southern shore of the Strait of Juan de Fuca (Figure 1). Aircraft-based observers record all bird species seen below the high tide line, but monitoring goals and data summaries emphasize certain alcid, diving duck, loon, and grebe species.

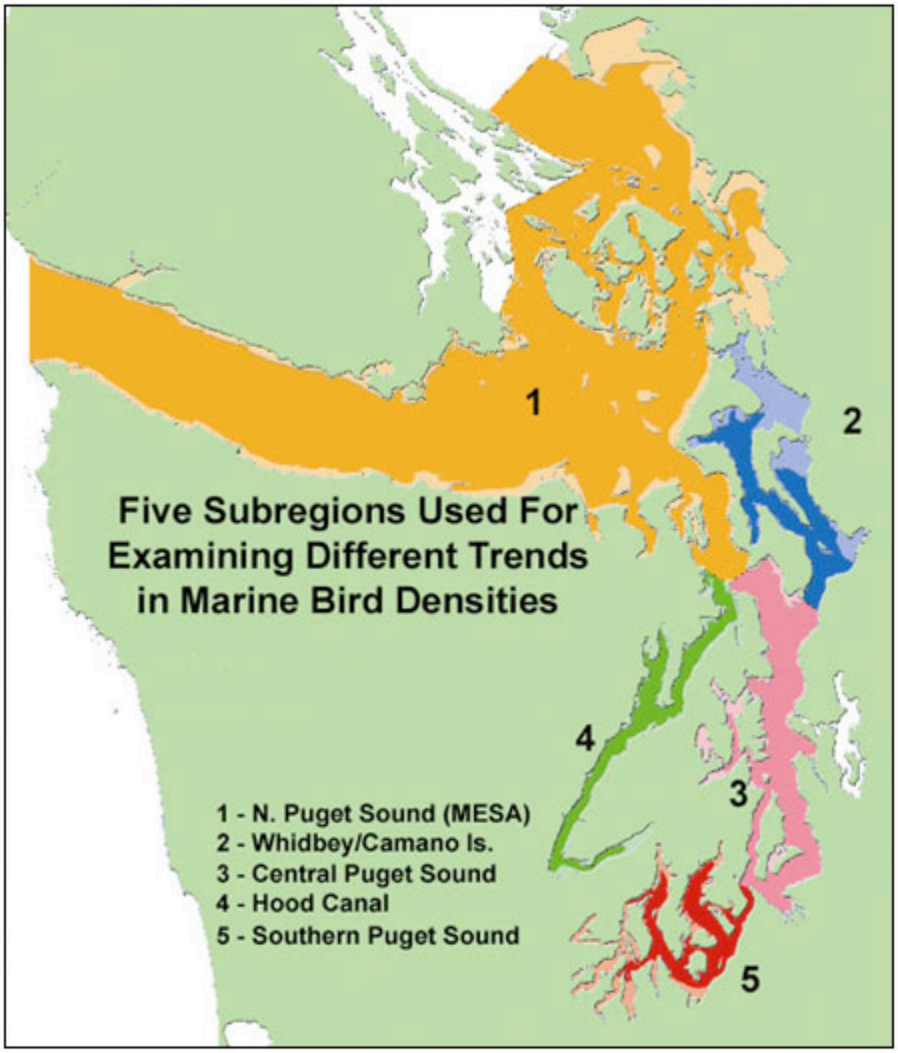


Figure 1. Map of PSAMP subregions (Reprinted from Evenson et al. 2010 with permission from Washington Department of Fish and Wildlife)

Nysewander et al. (2005) evaluated long-term changes in abundance in several species of marine birds by comparing the PSAMP results from 54 aerial transects with results from nearly identical MESA transects. Results of this analysis revealed significant declines in 13 of the 20 species or species groups studied, including declines in at least one species from each marine bird family found in northwestern Washington. For some species, such as the western grebe (*Aechmophorus occidentalis*) and long-tailed duck (*Clangula hyemalis*), declines were as high as 95% and 91%, respectively. Although methodologies used in MESA and PSAMP surveys were relatively comparable, differences did exist, for example in the locations and habitat types surveyed by the aerial methods, and in the proportion of the MESA baseline data that was from aerial, land-based and ferry-based surveys. Furthermore, the PSAMP used aerial surveys, but the potential bias associated with avoidance of aircraft by birds has not been evaluated.

Results from Nysewander et al. (2005) and other studies (e.g. Wahl 2002) sparked concern over declines in marine birds in Puget Sound. In acknowledgment of these concerns and the multiple problems associated with comparing results across disparate survey methodologies, the Western Washington University (WWU) or WWU/MESA comparison study was initiated (Bower 2009). The WWU/MESA comparison study replicated land-based and ferry-based transect portions of the MESA surveys over two winters (2003-2004 and 2004-2005). Results of the WWU/MESA comparison of data were largely consistent with the MESA/PSAMP comparison (Nysewander et al. (2005), although a few results diverged. To perform a third comparison of marine bird observations over time, Bower (2009) analyzed annual Christmas Bird Count (CBC) data from 11 count circles north of Puget Sound (1975-1984 and 1998-2007). Table 1 summarizes characteristics of the data sets used by to make comparisons (Bower (2009).

Table 1. Attributes of bird surveys compared in Bower (2009)

Study	Year(s)	Geographic area	Methods
Marine Ecosystems Analysis (MESA)	Jan-Dec 1978-1979	Admiralty Inlet (S), Tsawwassen-Schwartz Bay BC Ferry (N), Neah Bay (W), and WA mainland (E)	Shore-based point counts, ferry and small boat transects, aerial transects
Puget Sound Ambient Monitoring Program (PSAMP)	Winter 1992-1999	Straight coastline between Admiralty Inlet (S), Strait of Georgia (N), Neah Bay (W), and WA mainland (E)	Aerial transects compared with 1970s MESA aerial transects
Western Washington University (WWU)	Sept-May 2003-2004 and Sept-May 2004-2005	Admiralty Inlet (S), Tsawwassen-Schwartz Bay BC Ferry (N), Sand Juan Islands (W), WA mainland (E)	Shore-based point counts and ferry transects compared with 1970s MESA shore-based point counts and ferry transects
Christmas Bird Count (CBC)	1975-1984 and 1998-2007	Salish Sea, including 8 BC and 3 WA CBC circles	Standard CBC methods for 11 CBC circles, data from 1975-1984 with data from 1998-2007

Bower (2009) reported a 29% decline in the total number of marine birds in inland waters of the Salish Sea – which includes areas and data from outside the Puget Sound Basin – between 1978/79 and 2003-2005 (Figure 2). It should be noted, however, that this overall decline can be substantially influenced by changes exhibited by certain individual species. For example, of the 37 most common overwintering marine species, 14 showed significant declines and six showed significant increases. Notably, the largest declines were observed among taxonomically diverse groups, including common murre (*Uria aalge*) (–92.4%), western grebe (–81.3%), surf scoter (*Melanitta perspicillata*) (–59.8%) and brant (*Branta benicla*) (–73.2%). Species that showed increases in abundance included double-crested cormorant (*Phalacrocorax auritus*) (+97.7%) and pigeon guillemot (*Cepphus columba*) (+108.9%). Results from the CBC data comparison revealed significant declines in seven of the 37 most common species or species groups, with significant increases in three species (Bower 2009).

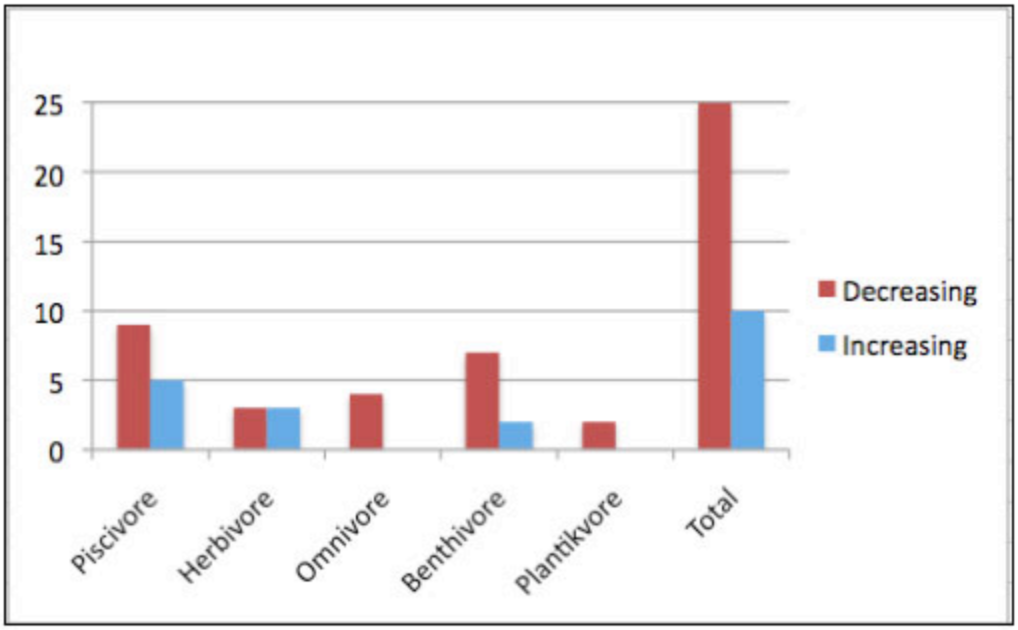


Figure 2. Changes in mean abundance among feeding guilds for 35 common overwintering marine birds in the Salish Sea between 1978/79 and 2003-2005 (data from Bower 2009)

Seventeen species or species groups were common to all three studies (the WWU/MESA comparison, the PSAMP/MESA comparison, and the CBC data comparison (Bower 2009). The PSAMP/MESA comparison revealed declines in more species (14 of 17) than did either the WWU/MESA comparison (six of 17) or the CBC comparison (three of 17) (Table 2). The PSAMP/MESA comparison showed no change or an increase in just three of 17 species or species groups, whereas the WWU/MESA comparison found no change or an increase in six of 17 and the CBC comparison found no change or an increase in eight of 17 species or species groups. Despite these differences, several consistencies emerge. First, the number of species declining exceeded the number of species increasing in all three comparisons. Second, three taxa -- western grebe, all scaup, and marbled murrelet-- showed declines across all three studies. And finally, only two species (Harlequin Duck, Bald Eagle) showed significant increases across all the three comparisons (Bower 2009).

Table 2. Comparison of percent change detected in three studies of non-breeding marine bird abundance for 17 species or species groups in Puget Sound (Bower 2009)

	Studies		
Species	PSAMP/MESA	WWU/MESA	Recent/historic CBCs

Common Loon	-64a		+49a		+13	
All loons	-79a			-33		-47
Red-necked Grebe	-89a		-46a			-35
Horned Grebe	-82a		-72a			-30
Western Grebe	-95a		-81a		-86a	
Double-crested Cormorant	-62a		+98a		+171a	
All cormorants	-53a		-8.3a			-25
Great Blue Heron		-19	+51			-16
Brant	-66a			-73	+1027a	
All scaup	-72a		-65a		-51a	
Harlequin Duck	+189a		+20		+7	
Long-tailed Duck	-91a			-44	+49	
All scoters	-57a		-33a			-8
Bufflehead	+20			-11	+5	
Bald Eagle	+35		+187a		+28	
Pigeon Guillemot	-55a		+109a		+15	
Marbled Murrelet	-96a		-71a		-69a	

a Statistically significant

In summary, widespread changes in the abundance of marine birds during the non-breeding season have occurred over the last 30 years in the Salish Sea (Nysewander et al. 2005, Bower 2009). Causes of these declines are not adequately known.

Scoters

Puget Sound supports some of the largest wintering scoter populations on the west coast of North America (Wahl 1981), where they feed on regionally-abundant bivalves and forage fish roe. Puget Sound is also one of the three most important staging areas and one of two major molting areas for other west coast scoter populations, including scoters that winter in California, Mexico, and British Columbia. Scoter populations in Puget Sound, including the wintering, staging, and molting populations, consist primarily of surf scoters and white-winged scoters (*M. fusca*). Black scoters (*M. perspicillata*) are also present, but in much smaller numbers. Surf scoters are one of the most abundant diving ducks in Puget Sound between September and May, with the highest densities in southern and central Puget Sound (Nysewander et al. 2005). Washington’s wintering scoters spend eight to 10 months in marine waters, with males spending approximately a month longer than females, before migrating to the Canadian interior to breed on freshwater lakes.

Scoters in Puget Sound are found most often in shallow coastal waters (< 20 meters depth) over a broad range of substrates, including pebbles, sand, mud, cobble, and rock. Previously thought to subsist on a relatively narrow diet of bivalves, scoters are now understood to adjust foraging patterns and locations to take advantage of ephemeral food sources. During much of the winter, they forage on newly-settled mussels and soft substrates inhabited by clams and other shellfish. In spring, some scoters in the region feed on herring eggs where available and flocks of surf scoters regularly track the northward progression of spawning events to consume this abundant and energy-rich source of food (Vermeer 1981). Anderson et al. (2008) found that prey such as crustaceans and polychaetes associated with eelgrass habitats comprise a substantial part of scoter diets in late summer.

Scoters observed in both nearshore and offshore waters during PSAMP winter monitoring efforts between 1992 and 2008 ranged in mean overall densities from 9.2 to 19.4 birds per km2 (Evenson et al. 2010). The density indices reported for nearshore areas, which scoters favor, ranged from 34.8 to 70.4 birds per km2. Figure 3 shows scoter densities between 1992 and 2008. Of all scoters counted along transects sampled during 1992-2008 winter surveys, between 33% and 90% were identified to species in any single year. Of these, surf scoters comprised 55-82%, white-winged scoters comprised 17-40%, and black scoters made up 0.2-9%. WDFW currently is conducting species/age/sex ratio surveys by boat to provide a better estimate of species proportions (Evenson 2010).

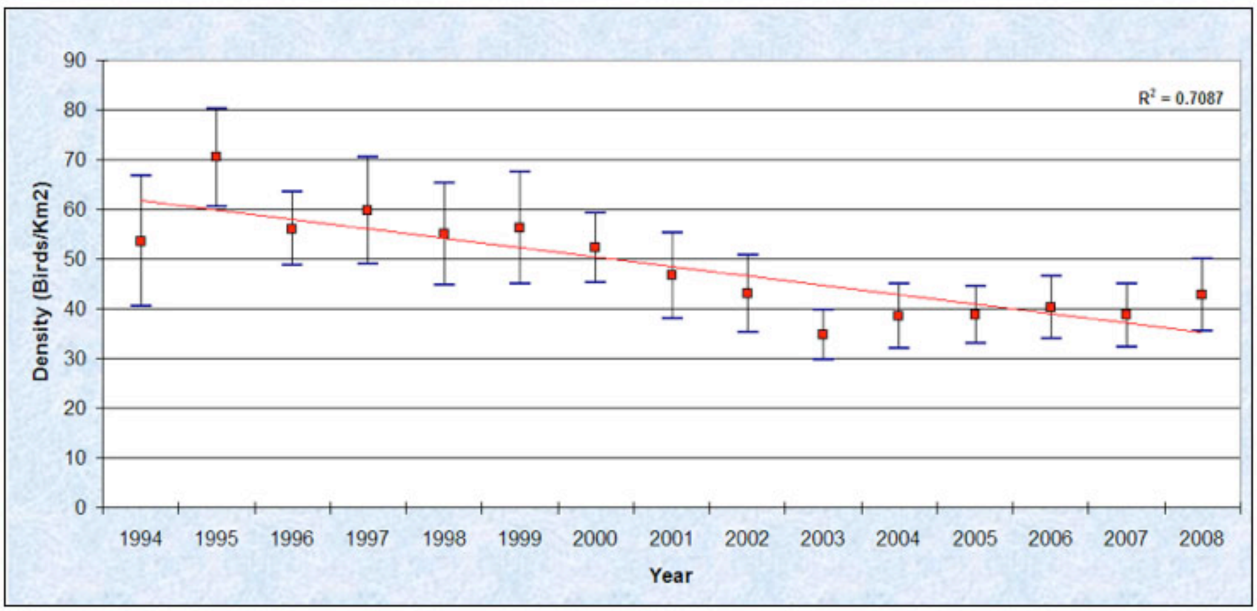


Figure 3. Mean winter densities of scoters in nearshore (<20 m) habitats of the inner marine waters of Washington state, 1993-2008 (Reprinted from Evenson et al. 2010 with permission from Washington Department of Fish and Wildlife)

Bower (2009) demonstrated that as a group scoters showed significant declines in both the PSAMP/MESA (-57%) and WWU/MESA (-33%) comparative studies. Surf scoters declined by 60% in the WWU/MESA comparison; however, nearly half of this decline is attributed to the collapse of the Cherry Point herring stock that occurred between the two survey periods (Stout 2001, Bower 2009). The evidence for this decline is compelling: more than 40,000 surf scoters were observed by MESA researchers in 1978 and less than 1,000 surf scoters were seen by WWU researchers at the same location in 2004 and 2005.

Comparisons of annual changes in density in the inner marine waters of Washington between 1992 and 2008 suggest that the scoters declined from the early 1990s through 2003, but that since 2003, densities have been relatively stable (Figure 3)(Evenson et al. 2010). However, spatial variation in rates of decline exist, for example in the Whidbey/Camano (North Puget Sound)area, where scoter densities have continued to decline (Figure 4)(Evenson et al. 2010). In 1993, the densities in the Whidbey/Camano area were the highest in the inner marine waters of Washington, but by 2008 densities in the Whidbey/Camano were the lowest (Figure 4)(Evenson et al. 2010).

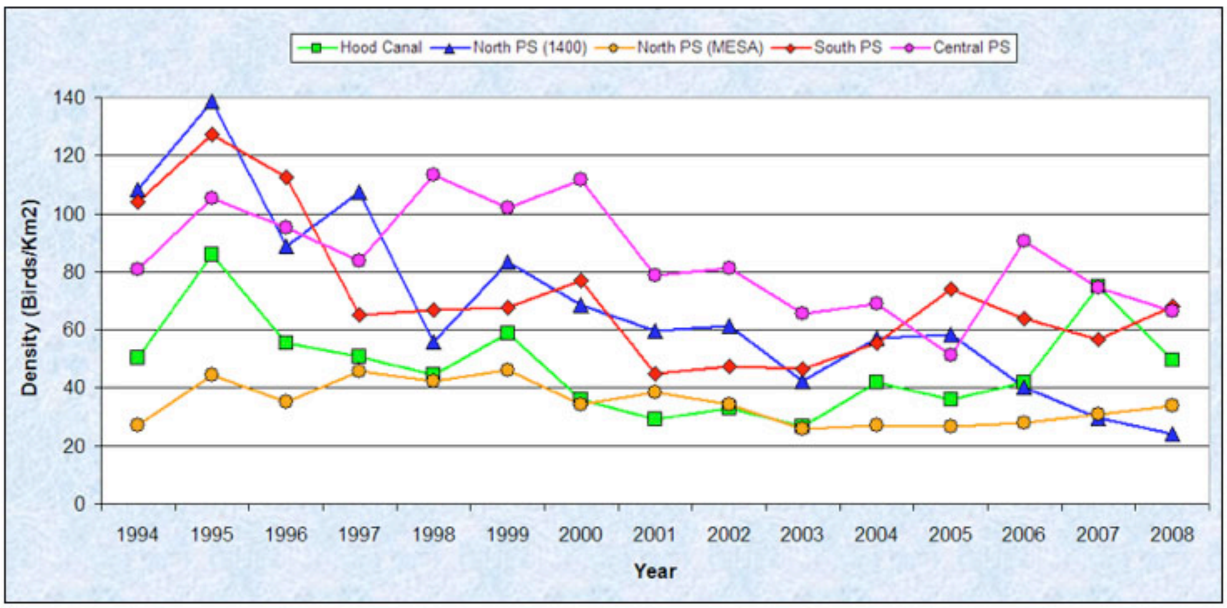


Figure 4. Comparison of winter scoter densities by region in the nearshore (<20 m) inner marine waters of Washington state (Puget Sound), 1993-2008 (Reprinted from Evenson et al. 2010 with permission from Washington Department of Fish and Wildlife)

Loons and Grebes

Several species of loons and grebes spend a substantial portion of the winter in Puget Sound where they utilize a variety of marine habitats. Loon species include the common loon (*Gavia immer*), Pacific loon (*G. pacifica*), and red-throated loon (*G. stellata*). Common loons are widespread and fairly common during winter in almost all nearshore marine habitats, and in most freshwater habitats, except rivers, typically occurring as single birds or in small numbers. They are rare breeders in Washington waters with the majority nesting throughout Canada and Alaska. Common loons were listed as sensitive by WDFW because they are a rare breeding species in the state and are vulnerable to a number of threats, including destruction or alteration of nesting habitat, poor water quality (i.e., degradation of lakes), and human activity (Richardson et al.

2000). Pacific loons are also widespread and common during winter, but occur further offshore than common loons and are more likely to congregate. Flocks of Pacific loons feed on schools of small fish near banks, tidal rips, and other hydrographic features of deeper waters. This species breeds in eastern Siberia and from northern coastal Alaska across to Baffin Island and Hudson Bay in North America. Red-throated loons are widespread and fairly common during winter in Puget Sound; they breed throughout Alaska, Canada, Greenland, and northern Europe Asia, with the very southern portion of their range extending south to southern Vancouver Island. Although red-throated loons can frequent many different types of marine waters, they tend to favor estuaries and shallow offshore areas, aggregating at times in areas where prey species are concentrated by tidal conditions.

Western grebes utilize marine and fresh waters in Puget Sound between October and April and tend to occur in groups. The primary wintering habitat for the larger flocks of western grebes are in offshore (>20m depth) marine waters with minimal tidal current flow, where they prey on schooling forage fish, although they may also occur in many saltwater situations and on inland lakes. Western grebes gather in large resting groups during the daytime hours and then disperse at night to forage. Major concentration areas have been identified through PSAMP aerial surveys (Evenson et al. 2010). Western grebes breed from southern British Columbia and the prairie Provinces in Canada south to Mexico.

Comparisons of survey data (Nysewander et al. 2005, Bower 2009) reveal that Puget Sound loon and grebe species have declined significantly in recent decades. Bower (2009) detected declines in loons as a group in all three comparative studies as follows: 64% decline in MESA/PSAMP comparison; 33% decline in WWU/MESA comparison; and 47% decline in historic/recent CBC comparison (Table 2). Records from the annual PSAMP winter aerial surveys from 1992 to 2008 show that loons constituted 0.8% of all marine birds surveyed (Evenson et al. 2010). The majority of loons were identified to species (common loon [28%], Pacific loon [27.9%], and red-throated loon [32.5%]) and occurred in both nearshore and offshore waters.

Among the three loon species, densities were lowest in the common loon, ranging from 0.17 to 0.57 birds per km2. A comparative analysis of common loon densities reported in MESA (1978/79) and PSAMP (1992-1999) surveys showed a 64% decline (Nysewander et al. 2005). Conversely, Bower (2009) reported 49% and 13% increases as shown by WWU/MESA and the historic/recent CBC data comparisons, respectively, which include survey data through the mid-2000s. It is unclear whether these changes reflect some degree of recovery since 1999, shifts in distribution, or are an artifact of differing or variously effective methodologies (see Uncertainties section, below).

Densities of Pacific loons observed during PSAMP winter surveys ranged from 0.26 to 1.21 birds per km2 in 1994-2008, with higher densities (10 and 89 birds per km2) observed in areas where flocks concentrate. Pacific loon winter densities appeared to be relatively stable over the period 1994-2008, although this result may be confounded by other loon species (Evenson et al. 2010). A comparison between MESA and PSAMP data was not made for Pacific loons due to the difficulty of distinguishing Pacific loons from red-throated loons in aerial surveys. Analysis of PSAMP subregional density indices suggest that Pacific loons favor certain subregions, such as northern Puget Sound, Whidbey/Camano Islands, and Central Puget Sound near Bainbridge Island (Evenson et al. 2010).

Red-throated loon densities of 0.17 to 1.20 birds per km2 were observed during PSAMP winter surveys of nearshore and offshore areas between 1994-2008 (Evenson et al. 2010). Densities appear to have been relatively stable over the past two decades (Evenson et al. 2010), although this species is not clearly separated from other loon species in some survey data.

Grebes

All grebe species wintering in Washington marine waters have exhibited some degree of decline over the last two decades, but western grebes have declined most sharply (Evenson et al. 2010). Overall densities for western grebes, combined for both nearshore and offshore waters, ranged from 3.9 to 13.2 birds per km2, while densities in the vicinities of the flocks ranged from 50 to 1,343 birds per km2 (Figure 5). A comparative analysis of western grebe densities reported by MESA (1978/79) and PSAMP (1992-1999) surveys showed a 95% decline (Nysewander et al. 2005). Bower (2009) noted that declines were observed in all three comparative studies (Table 2), across of the Salish Sea, and in every month of the MESA/WWU comparative surveys. Density indices reported by PSAMP winter monitoring surveys between 1992-2008 suggest that this species is still declining (Evenson et al. 2010).

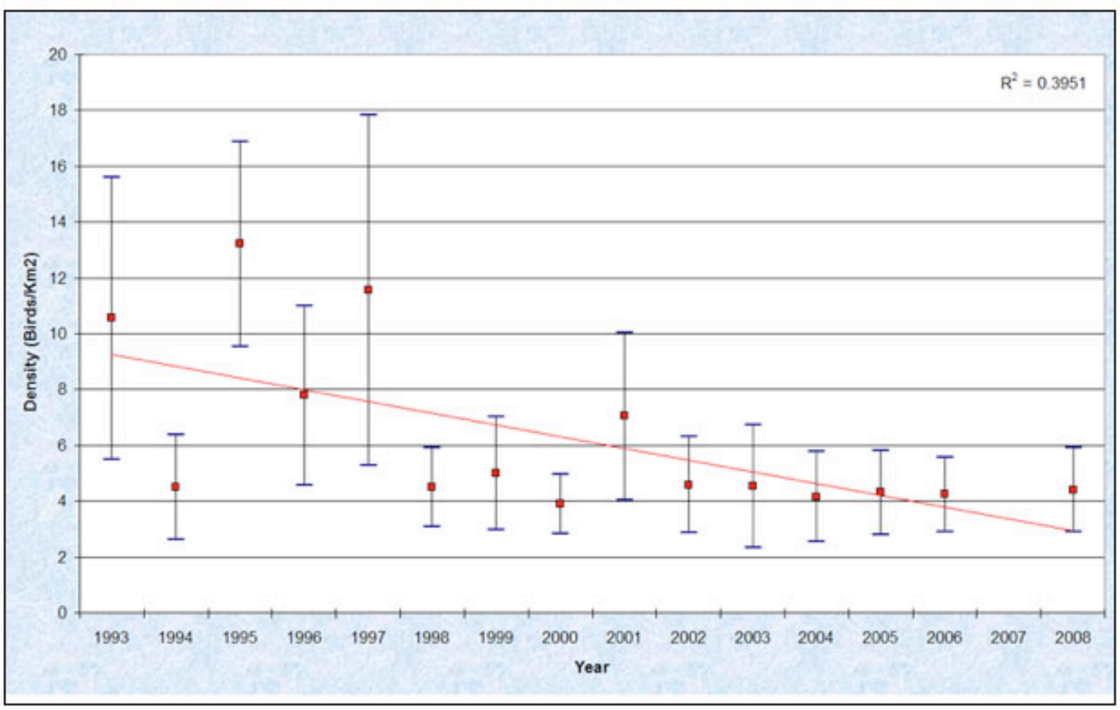


Figure 5. Winter Trends in Western Grebe Densities in the Inner Marine Waters of Washington State, 1993-2008 (Reprinted from Evenson et al. 2010 with permission from Washington Department of Fish and Wildlife)

Alcids

Several alcid species utilize marine waters of Puget Sound during winter months, with some species breeding along coastlines and on islands. Pigeon guillemots (*Cepphus columba*) are common and widespread residents that feed in nearshore habitats, along rocky shorelines, passes, banks, areas with tidal currents and rips, as well as in shallow embayments. These birds are seldom seen in flocks, except near colonies during breeding, although they can aggregate in productive feeding areas such as tidal convergences and passes. Pigeon guillemot nest in nearly every small-island or saltwater coastline habitat, with larger colonies are found in San Juan, Jefferson, Island, and Clallam Counties. Smaller colonies and single pairs are found throughout Puget Sound, making them the second most common breeding seabird in Puget Sound.

Rhinoceros auklets (*Cerorhinca monocerata*) are found throughout Puget Sound in both coastal habitats and far from land. Most often they often feed close to shore, especially where tidal currents near islands create localized upwelling and trophic intensification. Flocks may overnight in protected bays and forage farther out to sea during the day. Rhinoceros auklets in Washington nest at three main sites: Destruction Island, Protection Island, and Smith Island. Smaller numbers nest at a few other sites in Puget Sound.

Marbled murrelets (*Brachyramphus marmoratus*) are small, fast-flying seabirds present year round in coastal areas throughout Washington. They are non-colonial alcids that breed in mature inland forests up to 84 km from marine shorelines that support prey such as small schooling fish or invertebrates in shallow waters (Raphael 2006). Areas of winter concentration in Washington include the southern and eastern end of the Strait of Juan de Fuca, Sequim, Discovery and Chuckanut Bays, and the San Juan Archipelago. In 1992, the Pacific coast population of marbled murrelets south of the Canadian border was listed as Threatened by both USFWS and the State of Washington. Critical habitat in Washington, Oregon and California was designated in 1996. Primary threats to marbled murrelets include the loss and modification of nesting habitat, primarily due to commercial timber harvesting of older forests, effects resulting from oil spill pollution, and to a much lesser degree, risks associated with capture in commercial fisheries gear (Ralph et al. 1995).

In 2003, a WDFW survey of pigeon guillemot colonies in Puget Sound reported at least 471 colonies, representing approximately 16,000 breeding birds (Evenson et al. 2003). Long-term changes in pigeon guillemot populations are not known due to absence of historical data. Records from annual PSAMP aerial surveys show that pigeon guillemot densities were highest in nearshore habitats (<20m depth), where they ranged from 0.26 to 1.18 birds per km2 in 1992-2008 (Evenson et al. 2010). Densities from the inner marine waters of Washington during winters 1993-2008 increased from 1993-1997, and then remained stable through 2008. A comparative analysis of pigeon guillemot densities recorded by MESA (1978/79) and PSAMP (1992-1999) surveys showed a 56% decline over that period (Nysewander et al. 2005). However, Bower (2009) reported a 109% increase in pigeon guillemot density based on the WWU/MESA comparative study, which covered a slightly longer time period (Table 2). The inconsistencies likely reflect differences in sampling between the studies (Bower 2009) and a the lack of knowledge of pigeon guillemot post-breeding dispersal patterns (Evenson et al. 2010).

Rhinoceros auklet breeding populations in Puget Sound are concentrated on Protection Island and Smith Island. Protection Island hosts 70% of the breeding birds within Washington's inner marine waters (Speich et al. 1989). Estimates of the breeding population of Rhinoceros auklets on Protection Island have shown a 30% decline in breeding pairs with more than 17,000 breeding pairs in 1975 (Wilson and Manuwal 1986) decreasing to approximately 12,000 pairs in 2000 (Wilson 2005).

In 2006, marbled murrelet population size was estimated to be about 22,000 in Washington, Oregon, and California (Huff et al. 2006), compared with approximately 860,000 in Alaska and 55,000 to 78,000 in British Columbia in 2004 (McShane 2004). At-sea counts of marbled murrelets using boat-based observer transects were conducted from 2000 to 2009 as part of effectiveness monitoring of the Northwest Forest Plan. In 2009, USFWS conducted a five-year status review of the Northwest Forest Plan and determined that marbled murrelets in Puget Sound had continued to decline significantly since the previous review conducted in 2002 (Pearson et al. 2010). The population estimate for marbled murrelets in all zones in the Northwest Forest Plan area (Washington, Oregon and California) was 17,791 (95% confidence interval: 14,631 – 20,952). Estimates from the 9 years of monitoring have ranged from 17,354 to 23,673. The 2009 population estimate for Puget Sound and Juan de Fuca Strait east of Cape Flattery from at-sea surveys was 5,623 birds (95% confidence interval: 3,922 – 8,352 birds). The annual rate of decline for the 2001-2009 period was 7.0% (standard error = 1.8%; Pearson et al. 2010). For Washington State overall, there was a significant decline in murrelet density for the 2001-2009 period (Pearson et al. 2010). The largest concentrations of birds occurred in northern Puget Sound and the Strait of Juan de Fuca.

High Arctic Black Brant

High arctic black brant are a subpopulation of brant geese that utilize Puget Sound shallow bays and saltwater marshes from late November through May. They breed in the high arctic of western Canada, primarily on Melville Island and Prince Patrick Island, and then stage for over a month at Izembek Lagoon in Alaska before heading to wintering grounds in Puget Sound. Brant wintering habitats are usually characterized by an abundance of eelgrass and marine algae (e.g., Padilla, Samish and Fidalgo Bays in Skagit County). Large concentrations of brant may gather at Dungeness Spit and Willapa Bay, but smaller flocks are present in the southern Puget Sound. Because of their strong dependence on certain plants, fidelity to wintering and breeding locations, and because some populations live in harsh environments, brant are more vulnerable to periodic breeding failures and occasional heavy losses from starvation than are most other geese (Reed et al. 1998).

Results from the comparative MESA and PSAMP studies showed that brant abundance varied widely over spatial and temporal scales (Bower 2009). Brant exhibited declines in the PSAMP/MESA comparision (-66%) and WWU/MESA comparison (-73.2%), but increased by more than 1000% in the CBC comparison data (Bower 2009). The large decline in the WWU/MESA comparison was principally driven by a decline in numbers on the primary wintering grounds of Padilla and Samish Bay. Outside these two locations, brant numbers showed a slight increase. CBC comparison data showed increases in brant in British Columbia, possibly indicating a change in the wintering location of brant.

Great Blue Heron

In Puget Sound, great blue herons (*Ardea herodias*) belong to a non-migratory and marine-oriented subspecies (*A. herodias* subsp. *fannini*) that ranges from Alaska to southern Washington state, with the largest concentration occurring in northwestern Washington and southwest British Columbia (Butler 1997). During the non-breeding season, great blue herons are widely dispersed in Puget Sound, utilizing coastal and lowland areas for foraging and roosting. They are often found as solitary individuals. In contrast, between late winter and summer, herons occur in high densities centered on nesting colonies and associated foraging sites. Herons forage in a variety of habitat types depending on local conditions, tides, and season. Saltwater and freshwater marshes provide year-round foraging opportunities of fish, crustaceans, amphibians and reptiles, though terrestrial habitats also provide small mammals in heron diets (Eissinger 2007).

Marine shoreline and intertidal areas are important to the success of coastal heron colonies. In 2004, WDFW performed an aerial survey to determine foraging habitat, distribution, and concentration areas of great blue herons in Puget Sound (Hayes 2006). Based on this survey it was estimated that 73% of the active heron colonies in Puget Sound are directly associated with marine and estuarine intertidal habitats for foraging activities during the breeding season. In particular, the reproductive success of colonies is dependent on prey associated with eelgrass habitats (Eissinger 2007), such as Drayton Harbor, Port Susan, and Samish, Padilla, and Skagit bays.

Few records of historical trends exist for the great blue heron in Puget Sound. Methods for monitoring heron colonies in British Columbia and Puget Sound have recently been developed, although they are not yet standardized between the two areas. In western Washington, colony status has been assessed approximately every four years by WDFW biologists, and larger colonies in certain locations are monitored by independent investigators or conservation groups. Eissinger (2007) conducted a review of available population data and concluded that since the mid-1990s, the population of northwestern great blue herons has been stable, with the current estimate at 4,700 nesting pairs or 9,400 breeding herons in 2003-2004 (Figure 6). This breeding population represents 121 colonies located on Vancouver Island, the British Columbia mainland, and in the Georgia Strait and Puget Sound basins. Notably, approximately 66% of the total population is concentrated in only 16 colonies, and 35% of the total breeding population belongs to five mega-colonies supporting 200-600 breeding pairs each. In the past decade, the Puget Sound population has seen a substantial transformation from a diffuse distribution of smaller colonies across the landscape to larger colonies in upland marine areas. Reasons for this shift are unknown, but possible causal factors include combinations of increased predation by expanding bald eagle population, human disturbance and encroachment on habitat, degradation and fragmentation of nearshore and coastal habitats by development and land use activities, pollution, changes in prey abundance or distribution, and other systemic changes related to ecosystem decline (Eissinger 2007).

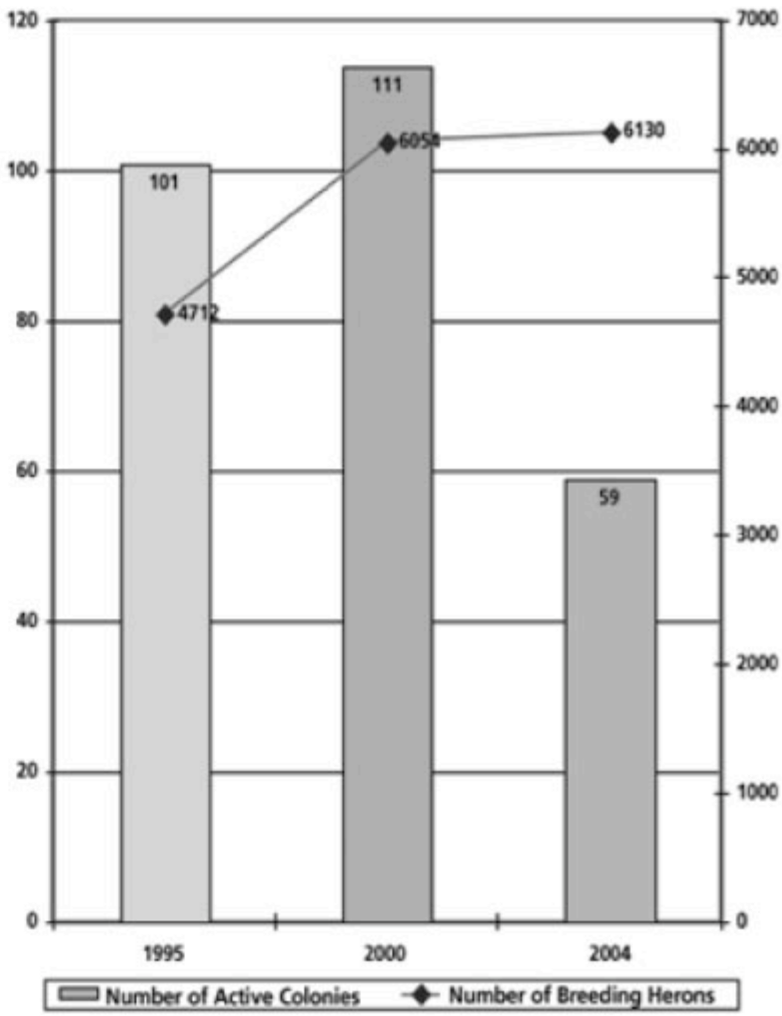


Figure 6. Great blue heron population trends in Puget Sound (reprinted from Eissinger 2007 with permission from the Puget Sound Nearshore Ecosystem Restoration Project and Washington Department of Fish and Wildlife)

Uncertainties

With the recovery of Bald Eagle populations, anecdotal information indicates predation pressure (direct and indirect) has increased at Great Blue Heron colonies. The effect of increasing Bald Eagle presence on colony persistence or productivity by Great Blue Herons is not known.

Trends in waterbird abundance derived from Christmas Bird Counts must be assessed to evaluate whether correction factors that account for observer effort (e.g. party hours) are appropriate. Correction factors applied where they are not necessary could result in a conclusion that abundance had decreased when in fact it had not changed.

Many marine birds migrate, overwinter or breed in regions quire distant from the area(s) they use in Puget Sound. The degree to which potentially significant limiting factors in those areas influence observed changes in abundance in Puget Sound is largely unknown.

Additional work is needed to determine whether changes in abundance of particular marine birds reflect actual population changes or shifts in regional distribution that would locally mimic population declines.

Summary

Multiple species of marine bird that overwinter in Puget Sound have shown sharp declines in abundance over the past two decades. Declining species outnumber increasing species, declines occur across diverse taxonomic groups and feeding guilds, and declines of up to 95% have been reported. Reasons for these declines are not well established and may include factors operating locally, along migration flyways, and at the breeding grounds. Habitat loss and changes in food availability or abundance may have contributed to population changes.

Literature Cited

Anderson, E. M., J. R. Lovvorn, and M. T. Wilson. 2008. Reevaluating Marine Diets of Surf and White-Winged Scoters: Interspecific Differences and the Importance of Soft-Bodied Prey. The Condor 110:285-295.

Bower, J. L. 2009. Changes in marine bird abundance in the Salish Sea: 1975 to 2007. Marine Ornithology 37:9-17.

Butler, R. W. 1997. The great blue heron : a natural history and ecology of a seashore sentinel. UBC Press, Vancouver, BC.

Eissinger, A. M. 2007. Great blue herons in Puget Sound. Puget Sound Nearshore Partnership, Olympia, WA.

Evenson, J. R. 2010. Analysis of marine bird data, Puget Sound Assessment and Monitoring Program. Seattle, WA.

Evenson, J.R, D.R. Nysewander, B.L. Murphie, and T.A. Cyra. 2010. Marine Bird and Mammal Component of the Puget Sound Ambient Monitoring Program web reports. Available at: http://wdfw.wa.gov/mapping/psamp/index.html. Washington Department of Fish and Wildlife, Olympia, WA.

Evenson, J. R., D. R. Nysewander, M. Mahaffy, B. L. Murphie, and T. A. Cyra. 2003. Status, abundance, and colony distribution of breeding pigeon guillemots (*Cepphus columba*) from the inland marine waters of Washington State, as documented by PSAMP efforts, 2000-2002 in Proceedings from Georgia Basin/Puget Sound Research Conference.

Hayes, G. 2006. Puget Sound Great Blue Heron Foraging Aerial Survey 2004. Washington Department of Fish and Wildlife, Olympia, WA.

Jewett, S. G. 1953. Birds of Washington State. University of Washington Press, Seattle.

McShane, C. 2004. Evaluation report for the 5-year status review of the marbled murrelet in Washington, Oregon, and California. EDAW, Inc., Seattle, WA.

Nysewander, D., J. Evenson, B. Murphie, and T. Cyra. 2005. Report of marine bird and mammal component, Puget Sound Ambient Monitoring Program, for July 1992 to December 1999 period. Washington Department of Fish and Wildlife, Olympia, WA.

Ralph, C. J., G. L. Hunt, Jr., M. G. Raphael, and J. F. Piatt. 1995. Ecology and conservation of the marbled murrelet. U.S. Dept. of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA.

Raphael, M. G. 2006. Conservation of the Marbled Murrelet under the Northwest Forest Plan. Conservation Biology 20:297-305.

Rathbun, S. F. 1915. List of Water and Shore Birds of the Puget Sound Region in the Vicinity of Seattle. The Auk 32:459-465.

Reed, A., D. H. Ward, D. V. Derksen, and J. S. Sedinger. 1998. Brant (*Branta bernicla*): Birds of North of America.in A. Poole, editor. The Birds of North America Online. Cornell Lab of Ornithology, Ithaca, NY.

Richardson, S. A., D. Hays, R. Spencer, and J. Stofel. 2000. Washington State status report for the common loon. Washington Dept. of Fish and Wildlife, Olympia, WA.

Speich, S. M., T. R. Wahl, and J. F. Watson. 1989. Catalog of Washington seabird colonies. U.S. Dept. of the Interior, Fish and Wildlife Service, Research and Development, Washington, D.C.

Stout, H. A. 2001. Status review of Pacific herring (*Clupea pallasi*) in Puget Sound, Washington. U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, WA.

Vermeer, K. 1981. Food and populations of Surf Scoters in British Columbia. Wildfowl 32:107-116.

Wahl, T. R. 1981. Marine bird populations of the Strait of Juan de Fuca, Strait of Georgia, and adjacent waters in 1978 and 1979. National Oceanic and Atmospheric Administration; United States Environmental Protection Agency, Research and Development, Seattle, WA; Washington, D.C.; Springfield, VA.

Wahl, T. R. 2002. Trends in numbers of marine birds wintering on Bellingham Bay. Washington Birds 8:29-40.

Wilson, U. W. 2005. The Effect of the 1997-1998 El Nino on Rhinoceros Auklets on Protection Island, Washington. Condor 107:462-468.

Wilson, U. W., and D. A. Manuwal. 1986. Breeding Biology of the Rhinoceros Auklet in Washington. Condor 88:143-155.

Bald eagles

Background

Bald eagles (*Haliaeetus leucophalus*) are present year-round throughout most parts of Washington with the highest densities in the Puget Sound region. Individuals occur in the Puget Sound basin as migrants, winter residents and members of the breeding population. They are often associated with shorelines and large, open expanses of water (Stalmaster 1987). Bald eagles are opportunistic foragers that feed most frequently on fish and waterfowl, and as both predators and kleptoparasites, possess a variety of hunting behaviors, consuming live fish and birds as well as scavenging upon dead fish (particularly salmonids), birds and mammals (Watson 2002, Stinson et al. 2007). They are known to hunt in both seabird (Kaiser 1989, Thompson 1989) and great blue heron colonies (Norman et al. 1989).

Breeding bald eagles require large trees near open water in locations that experience relatively low levels of human activity. In Washington, surveys by Washington Department of Fish and Wildlife (WDFW) conducted in 2005 showed that nearly all (97 %) of surveyed bald eagle nests were within 3,000 feet of shoreline (Stinson et al. 2007). While nests are most numerous near marine shorelines, many are also found on shores of lakes, reservoirs, and rivers of Washington. In a more detailed study of 53 breeding pairs throughout western Washington from 1986 - 1997, Watson et al. (2002) found that the mean home range size of 53 bald eagle pairs distributed across lakes, marine shorelines, rivers and bays was 4.9 km2, and ranged from approximately 2 to 7 km2. The density of nesting eagles depends on many factors that affect habitat quality including prey populations, degree of human disturbance, and the availability of nest and perch trees.

Breeding pairs initiate nesting activities in January or February and disperse by late summer when many migrate north to coastal British Columbia and southeast Alaska for several weeks to take advantage of food supplies associated with late summer and early fall salmon runs (Watson 1998). The timing of breeding activities in Washington has been summarized by Watson (2006). Fledglings also disperse northward, but they may remain there for several months before returning to Washington.

Washington's wintering eagles begin to arrive in October from northern breeding territories in Alaska and Canada. Most adults arrive in November and December and many juveniles arrive in January (Buehler 2000, Watson 2001). The winter distribution of bald eagles in Washington is similar to the breeding distribution, but more concentrated at salmon spawning streams and waterfowl wintering areas. Winter ranges are considerably larger and more variable than breeding ranges.

Threats to bald eagles include habitat degradation and reductions in prey such as salmonids in Puget Sound and its surrounding watershed. Alteration of upland nesting habitat from natural events (e.g., windstorms) or human-related factors (e.g., timber harvest, development) that results in either mortality or reduced availability of nest trees or suitable territories, can reduce the number of occupied nesting territories. Because average life expectancy of nests can be shorter than that of breeding birds (Stalmaster 1987), bald eagles often need trees of similar stature located nearby to serve as replacement nest trees if a nesting territory is to persist at the site.

Conservation Status

Bald eagles in Washington were listed as Threatened under the federal Endangered Species Act (ESA) in 1978. The widespread use of DDT between the 1940s and 1970s is widely viewed as the main cause of the decline of bald eagles in Washington and the other 48 states, though direct extirpation and habitat alteration are also known causes (Stalmaster 1987). In response to rebounding populations, the bald eagle was removed from protection under the ESA in 2007 (USFWS 2007a). The bald eagle is still protected by the Bald and Golden Eagle Protection Act and the Migratory Bird Treaty Act (USFWS 2007b). At the state level, bald eagles were down-listed to Sensitive status by the Washington Fish and Wildlife Commission. Habitat protection is still authorized in Washington by the Bald Eagle Protection Law of 1984 (RCW 77.12.655), which requires the establishment and enforcement of rules for buffer zones around bald eagle nest and roost sites. Habitat is protected through bald eagle management plans approved by WDFW. Between1986 and 2005, over 2,900 bald eagle plans were developed between WDFW and various landowner entities for activities on private, state, and municipal lands in Washington (Stinson et al. 2007).

Status

The most recent statewide breeding season census conducted by WDFW, in 2005, found 840 occupied nests in 1,125 territories searched (Stinson et al. 2007). Breeding activity was confirmed by the presence of eggs or shells in or around the nest or observations of adults incubating eggs or brooding chicks.

Trends

WDFW began localized monitoring of bald eagle nests in the San Juan Islands in the early 1960s. The first extensive survey that covered Washington’s entire marine shoreline was conducted in 1975 and statewide comprehensive activity and productivity surveys were conducted annually from 1980-1992. Nest activity surveys were continued through 1998, and conducted again in 2001 and 2005. From 1981 to 2005 the nesting population in Washington had increased seven fold (Figure 1)(Stinson et al. 2007). The number of bald eagle territories in Puget Sound also increased substantially (Figure 2)(Stinson et al. 2007). As of 2010, there were 751 known territories in the Puget Sound Basin, with most sites occurring in San Juan, Clallam, Island and Skagit counties (Table 1). Although historical estimates of the bald eagle population are not available, Stinson et al. (2007) estimated 1,328 serviceable breeding locations (SBL; analogous to a territory) existed in Washington prior to European settlement. We note, however, that the estimate of SBLs included various assumptions that cannot be evaluated relative to conditions of the 19th century. The number of known territories in 2010 was 1,403 (WDFW database), which suggests the population may be at or near carrying capacity. While the carrying capacity of bald eagles in Washington is not known, a recent decline in nest occupancy rate suggests that nesting habitat in parts of western Washington may be approaching saturation (Stinson et al. 2007). The number of resident breeders, and trends in localized winter counts suggest that Washington state hosts approximately 4,000 resident and migratory bald eagles each winter (Stinson et al. 2007).

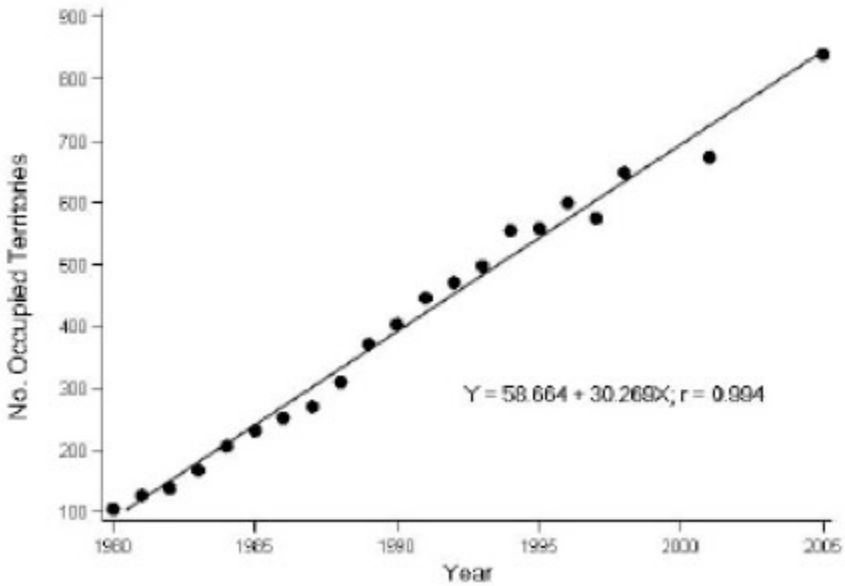


Figure 1. Time trend in population status (number of occupied nests), 1980-2005 (reprinted from Stinson et al. 2007 with permission from Washington Department of Fish and Wildlife)

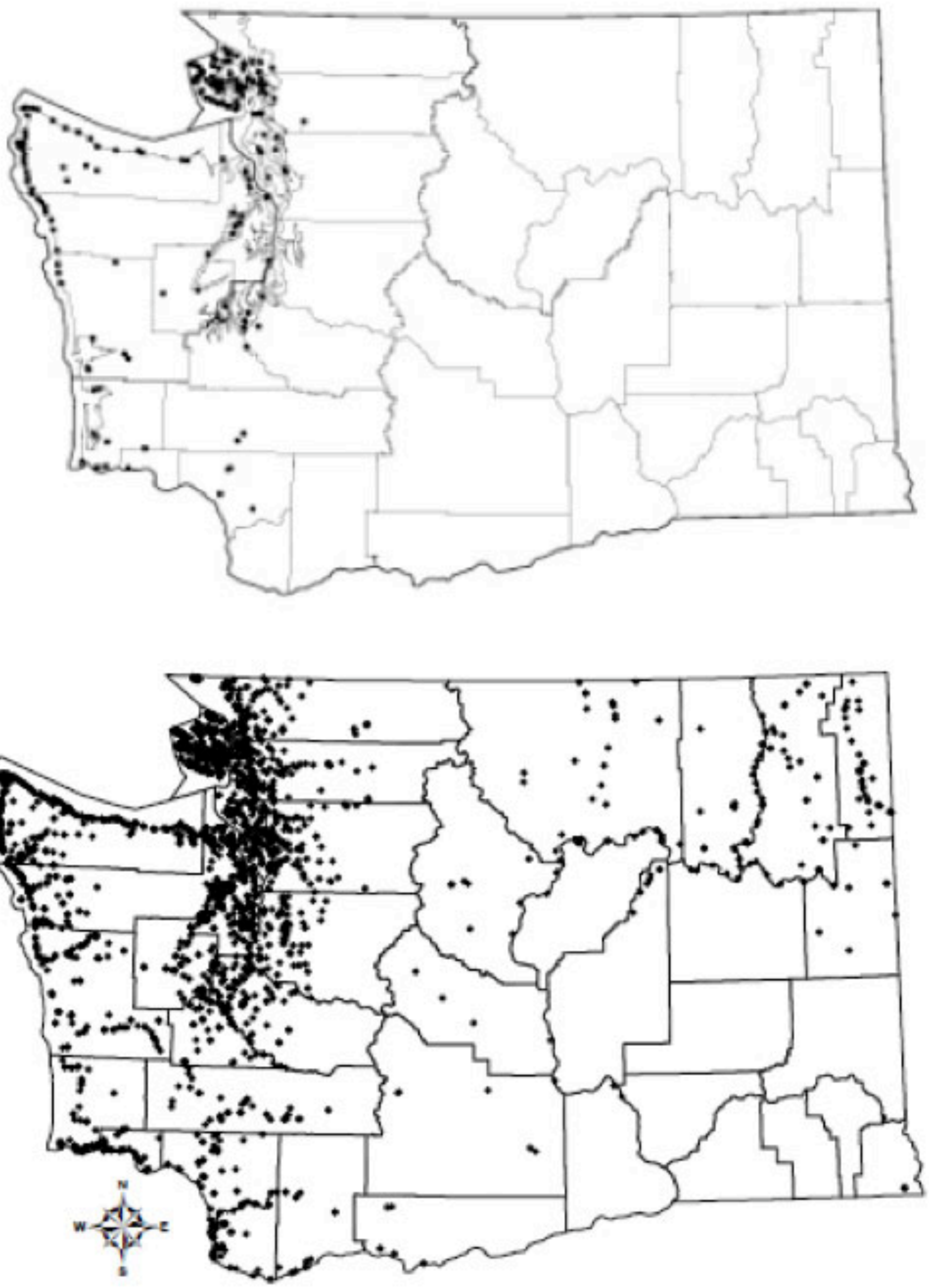


Figure 2. Distribution of bald eagle nests in Washington, in 1980 (top map) and 2005 (bottom map)(reprinted from Stinson et al. 2007 with permission from Washington Department of Fish and Wildlife)

Table 1. Number of bald eagle territories in those portions of Washington counties that are included in the Puget Sound basin. Data from WDFW database, methods according to Stinson et al.2007.

County	Number of Territories
Clallam	85
Island	84
Jefferson	59
King	51
Kitsap	69
Mason	33
Pierce	51
San Juan	98
Skagit	82
Snohomish	57
Thurston	17
Whatcom	65
Total	751

Uncertainties

1. The carrying capacity of bald eagles is unknown and likely varies from one ecosystem type or condition to another. Future monitoring will be necessary to identify carrying capacity.

2. Because bald eagles are closely associated with the marine environment, they are potentially vulnerable to contaminants in the marine food chain. The extent to which they may be vulnerable and the specific contaminant groups that might influence their physical or behavioral health are unknown.

3. The human population is expected to increase substantially in the next three decades. Much of the increase in Washington's population will likely occur in the Puget Basin. Potential responses to increased human pressures on habitats associated with nest territories, and the ability of existing rules to protect those habitats given increasing human pressures, are unknown.

Summary

Bald eagle abundance in Washington has increased in the past three decades, likely in response to federal and state management efforts. The number of nesting pairs in Washington is approximately eight times the number present when the use of DDT was restricted in 1972 (Stinson et al. 2007). In Puget Sound, the predicted rise in human population will continue to increase pressure on nesting and roosting habitats. State bald eagle protection rules (WAC 232-

12-292), along with other forest clearing regulations, may allow the population to persist at or near its current level.

Literature Cited

Buehler, D. A. 2000. Bald eagle (*Haliaeetus leucocephalus*) Birds of North America No. 506. 40 pp. Birds of North America, Philadelphia, PA.

Kaiser, G. W. 1989. Nightly concentration of bald eagles at an auklet colony. Northwestern Naturalist 70:12-13.

Norman, D. M., A. M. Breault, and I. E. Moul. 1989. Bald Eagle incursions and predation at great blue heron colonies. Colonial Waterbirds 12:215-217.

Stalmaster, M. V. 1987. The bald eagle. Universe Books, New York.

Stinson, D. W., J. W. Watson, and K. R. McAllister. 2007. Washington State status report for the bald eagle. Washington Dept. of Fish and Wildlife, Wildlife Program, Olympia, WA.

Thompson, S. P. 1989. Observations of bald eagles eating glaucous-winged gull eggs in western Washington. Northwestern Naturalist 70:13-14.

USFWS. 2007a. Removing the bald eagle in the Lower 48 states from the list of Endangered and Threatened wildlife: Final Rule. Federal Register 72: 37346-37372.

USFWS. 2007b. Authorization under the Bald and Golden Eagle Protection Act for take of eagles. Federal Register 72:31141-31155.

Watson, J. W. 2002. Comparative home ranges and food habits of bald eagles nesting in four aquatic habitats in western Washington. Northwestern Naturalist 83:101-108.

Watson, J.W. 2006. Bald eagle nesting chronology in western Washington. Washington Birds 9:8-11.

Watson, J. W., and D. J. Pierce. 1998. Ecology of bald eagles in western Washington with an emphasis on the effects of human activity. Washington Department of Fish and Wildlife, Olympia.

Watson, J. W., and D. J. Pierce. 2001. Skagit River bald eagles: movements, origins, and breeding population status. Washington Department of Fish and Wildlife, Olympia.

Harbor Seals

BackgrouNd

Harbor seals (*Phoca vitulina*) are found throughout temperate and arctic waters of the northern hemisphere, and inhabit coastal and estuarine waters along the eastern Pacific Ocean from Baja California north to the Gulf of Alaska and Bering Sea (Carretta et al. 2004, Carretta et al. 2007) . Harbor seals are found throughout the nearshore waters of Washington including Hood Canal, Puget Sound, the San Juan Islands, and the Strait of Juan de Fuca out to Cape Flattery (Jeffries et al. 2003) (Figure 1). They use hundreds of locations in Puget Sound to haul out or rest, including intertidal rocks, reefs, and beaches, logbooms, docks and floats. Harbor seals in Washington are considered non-migratory and display strong fidelity to haulout sites. Their local movements are associated with tidal cycles, time of day, weather, and prey availability (Zamon 2001, Carretta et al. 2004, Hayward et al. 2005, Carretta et al. 2007, Patterson and Acevedo-Gutierrez 2008). Most individuals in the inland waters forage in close proximity to haulout sites, and return to the same areas for foraging and haulout (Lance and Jeffries 2006). In general, harbor seals forage opportunistically on prey that are locally and seasonally abundant(Lance and Jeffries 2006, 2007).

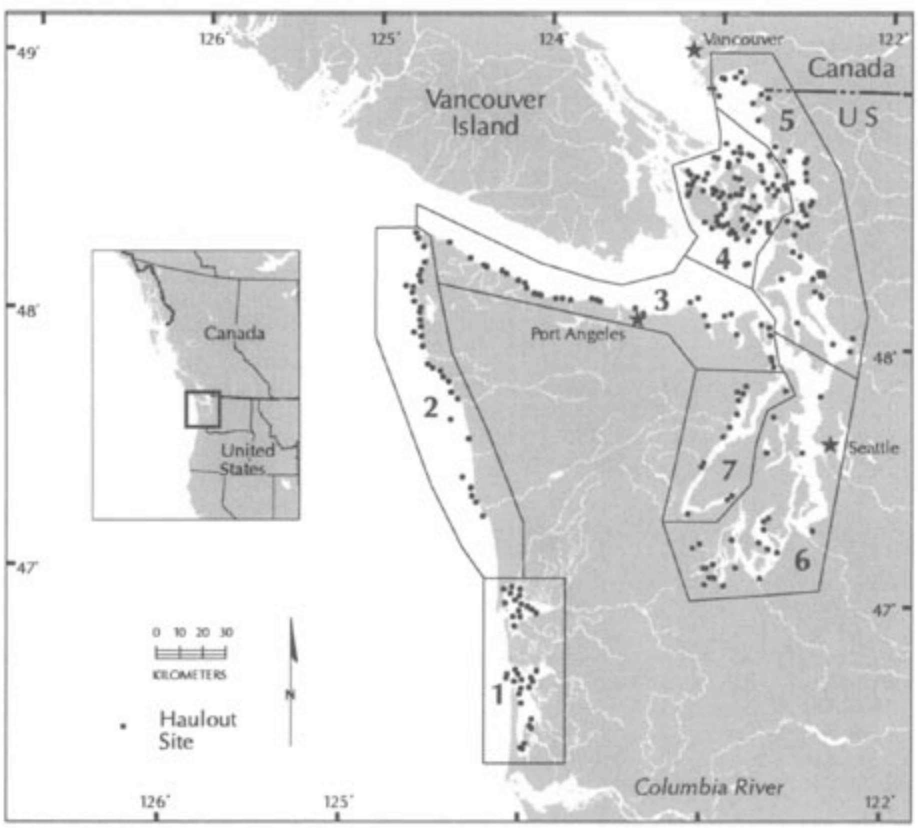


Figure 1. Map of harbor seal haulout sites and survey regions for Washington. The inland stock includes the Strait of Juan de Fuca (3), San Juan Islands (4), Eastern Bays (5), Puget Sound (6), and Hood Canal (7) (reprinted with permission from Jeffries et al. 2003).

Threats to harbor seals include incidental takes from drift gillnet fisheries, vessel strikes, and contaminants. Harbor seals are vulnerable to contamination by persistent organic pollutants (POPs) because they are long-lived, occupy a high trophic level, and have limited metabolic capacity to eliminate pollutants (Ross et al. 2004). Exposure to contaminants has also been associated with immunotoxicity and outbreaks of infectious disease (Mos et al. 2006). Harbor seals in Puget Sound are also heavily contaminated with polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs) (Simms et al. 2000, Ross et al. 2004, Cullon et al. 2005).

Status

Harbor seal numbers were severely reduced during the first half of the twentieth century by a state-financed population control program. This bounty program ceased in 1960, and in 1972, harbor seals became protected under the federal Marine Mammal Protection Act (MMPA) and by Washington State. Based on morphological, phenological and genetic differences, the coastal and inland populations of Washing are considered to be two different stocks (Carretta et al. 2007). Currently, both the inland and coastal stocks of harbor seals are not considered “depleted” under the MMPA or listed as “threatened” or “endangered” under the ESA. Population count data collected using aerial surveys of haulouts conducted by WDFW in 1999 indicate both stocks to be within their Optimum Sustainable Population (OSP) ranges as defined by Jeffries et al. (2003).

Trends

It is estimated that 2,000-3,000 harbor seals resided in Washington in the early 1970s (Newby 1973), and historic population levels prior to this are unknown. Beginning in 1983, WDFW initiated consistent aerial surveys of harbor seal inland waters population, which continued through 1999. Jeffries et al.(2003) found that during 1999, Washington inland stock contained 13, 692 seals and that both the coastal and inland populations were near carrying capacity (Figure 2). Thus, at the population levels of 1999, Jeffries et al. (2003) estimated that Washington State harbor seal populations could withstand significant declines and still be within the Optimum Sustainable Population levels. The 1999 population count continues to be the most recent estimate of Washington harbor seal abundances (Carretta et al. 2007).

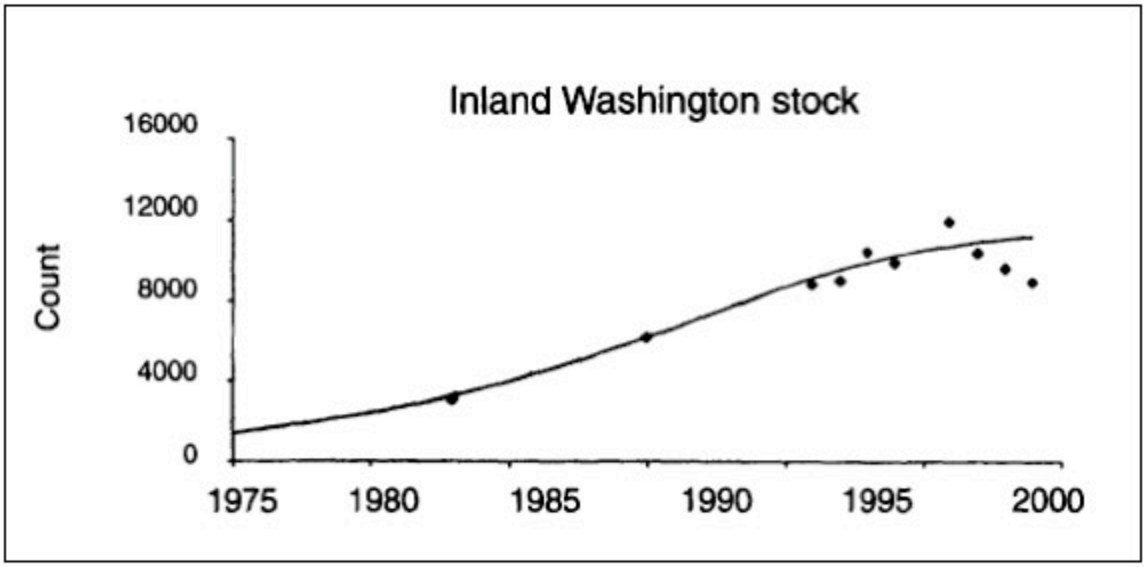


Figure 2. Generalized logistic growth curve of aerial counts of harbor seals in inland waters of Washington (includes the Strait of Juan de Fuca, East Bays, San Juan Islands, Hood Canal, and Puget Sound regions) (reprinted with permission from Jeffries et al. 2003).

Uncertainties

Harbor seal abundance estimates are based on aerial surveys of maximum haul-out counts, which can be complicated by spatial and temporal variability in the behavior of the seals and in the proportion of individuals that are observable (i.e., onshore) during sampling events. To address uncertainty in the proportion onshore, current estimates of trends and population abundances (Jeffries et al. 2003) use both a static correction factor developed by Huber et al. (2001) and an observation-error time series model fitting using maximum likelihood techniques to estimate population dynamic model parameters. To address variability in seal behavior, Hayward et al. (2005) suggest an environmentally dynamic modeling approach, but this has not been adopted. The impacts of contaminant exposure on population status are not well known.

Summary

Harbor seals populations in Washington State have recovered since the 1970s and population sizes may be near a stable equilibrium level, perhaps reflective of the current carrying capacity of the environment. Because of their high trophic position, harbor seal contaminant loads may be used as indicators of <u>pollution</u> levels in Puget Sound (Ross et al. 2004), and have been suggested as possible indicators of other types of anthropogenic change (climate change, fishing activities) (Hindell et al. 2003) and fish community composition (Lance and Jeffries 2007).

Literature Cited

Carretta, J. V., K. A. Forney, M. M. Muto, J. Barlow, J. Baker, B. Hanson, and M. Lowry. 2007. United States marine mammal stock assessments: 2006. National Marine Fisheries Service, La Jolla, CA.

Carretta, J. V., K. A. Forney, M. M. Muto, J. Barlow, J. Baker, and M. Lowry. 2004. U.S. Pacific Marine Mammal Stock Assessments: 2003. NOAA Technical Memorandum NMFS-SWFSC-358.

Cullon, D. L., S. J. Jeffries, and P. S. Ross. 2005. Persistent organic pollutants in the diet of harbor seals (*Phoca vitulina*) inhabiting Puget Sound, Washington (USA), and the Strait of Georgia, British Columbia (Canada): A food basket approach. Environmental Toxicology and Chemistry 24:2562-2572.

Hayward, J., S. M. Henson, C. J. Logan, C. R. Parris, M. W. Meyer, and B. Dennis. 2005. Predicting numbers of hauled-out harbour seals: a mathematical model. Journal of Applied Ecology 42:108-117.

Hindell, M. A., C. J. A. Bradshaw, R. G. Harcourt, and C. Guinet. 2003. Ecosystem monitoring: Are seals a potential tool for monitoring change in marine systems? Page 446 in N. Gales, M. Hindell, and R. Kirkwood, editors. Marine mammals: fisheries, tourism, and management issues. CSIRO, Collingwood, VIC.

Huber, H. R., S. J. Jeffries, R. F. Brown, R. L. DeLong, and G. VanBlaricom. 2001. Correcting aerial survey counts of harbor seals (*Phoca vitulina richardsi*) in Washington and Oregon. Marine Mammal Science 17:276-293.

Jeffries, S., H. Huber, J. Calambokidis, and J. Laake. 2003. Trends and status of harbor seals in Washington state: 1978-1999. Journal of Wildlife Management 67:207-218.

Lance, M. M., and S. J. Jeffries. 2006. Estimating importance of rockfish, lingcod and other bottomfish in the diet of harbor seals in the San Juan Islands. Page 20. Contract Report to SeaDoc Society, Olympia.

Lance, M. M., and S. J. Jeffries. 2007. Temporal and spatial variability of harbor seal diet in the San Juan Island archipelago. Research Agreement No. K004431-25, Contract report to SEAdoc society research agreement NO. K004431-25. Washington Department of Fish and Wildlife, Olympia, WA.

Mos, L., B. Morsey, S. J. Jeffries, M. B. Yunker, S. Raverty, S. De Guise, and P. S. Ross. 2006. Chemical and biological pollution contribute to the immunological profiles of free-ranging harbor seals. Environmental Toxicology and Chemistry 25:3110-3117.

Newby, T. C. 1973. Changes in Washington State harbor seal population, 1942-1972. Murrelet 54:5-6.

Patterson, J., and A. Acevedo-Gutierrez. 2008. Tidal influence on the haul-out behavior of harbor seals (*Phoca vitulina*) at a site available at all tide levels. Northwestern Naturalist 89:17-23.

Ross, P. S., S. J. Jeffries, M. B. Yunker, R. F. Addison, M. G. Ikonomou, and J. C. Calambokidis. 2004. Harbor seals (*Phoca vitulina*) in British Columbia, Canada, and Washington State, USA, reveal a combination of local and global polychlorinated biphenyl, dioxin, and furan signals. Environmental Toxicology and Chemistry 23:157-165.

Simms, W., S. Jeffries, M. Ikonomou, and P. S. Ross. 2000. Contaminant-related disruption of vitamin A dynamics in free-ranging harbor seal (*Phoca vitulina*) pups from British Columbia, Canada, and Washington State, USA. Environmental Toxicology and Chemistry 19:2844-2849.

Zamon, J. E. 2001. Seal predation on salmon and forage fish schools as a function of tidal currents in the San Juan Islands, Washington, USA. Fisheries Oceanography 10:353-366.

Killer Whales

Background

Three distinct groups of killer whales (*Orcinus orca*) occupy the coastal waters of the northeastern Pacific. These groups—northern and southern residents, transients, and offshores—are distinguished by diet, behavior, morphology, and other characteristics. Among these, Southern Resident and transient killer whales commonly are found in Puget Sound. Northern residents and offshore killer whales rarely enter Puget Sound (Wiles 2004, Kriete 2007), and therefore are not described in detail here.

While the taxonomic status of north Pacific killer whales remains unresolved (summarized in Krahn et al. 2004, NMFS 2008), the Southern Resident killer whale (SRKW) and transient killer whale populations are considered by NOAA to be separate stocks based on genetic, morphological, dietary and behavioral differences and are classified as endangered (SRKW) and threatened (transient) under the U.S. Endangered Species Act (2005). The SRKW population is found primarily in Washington and southern British Columbia and includes three groups or pods (J-, K- and L-pod) (Krahn et al. 2002, Krahn et al. 2004). Their home range during the spring, summer, and fall includes Puget Sound, the Strait of Juan de Fuca, and the Strait of Georgia (NMFS 2008) (Figure 1). During the late fall to winter, SRKWs travel as far south as central California and north to the Queen Charlotte Islands, British Columbia. The distribution of transient killer whales ranges from southern California to Icy Strait and Glacier Bay in Alaska (Ford et al. 2000). Transients are recorded along the Puget Sound and Vancouver Island shorelines during the summer and early fall (Wiles 2004).

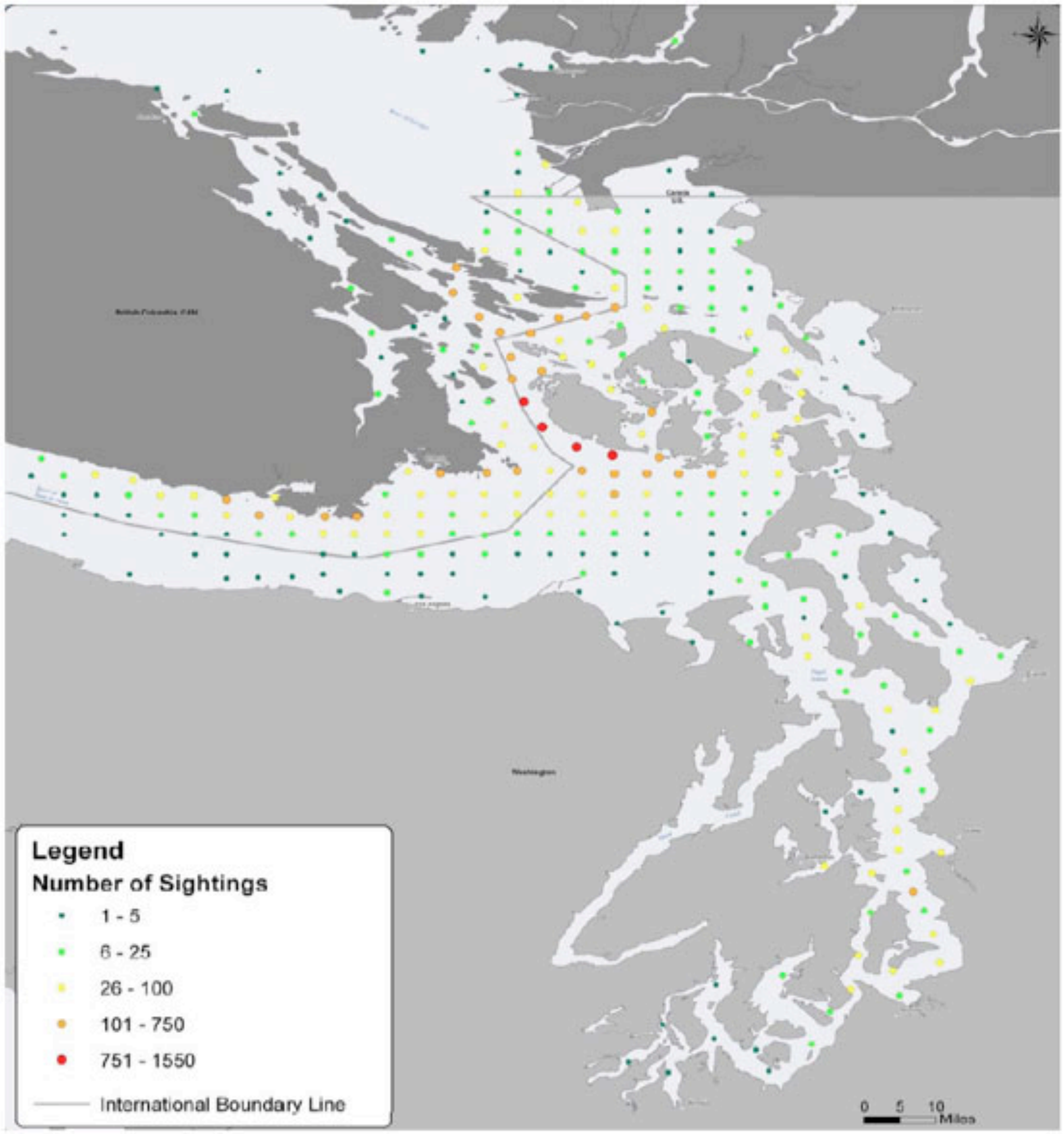


Figure 1. Distribution of Southern Resident killer whale sightings from 1990-2005 (data from The Whale Museum 2005; figure reprinted from NMFS 2008, courtesy of NOAA Fisheries).

Resident killer whales are believed to principally consume marine fish, while transients prey solely on marine mammals (Ford et al. 1998, Ford et al. 2000). Diets of resident killer whales were found to include 22 species of fish and one species of squid (Ford et al. 1998). A detailed dietary study based on 529 observed predation events from 1997 – 2005 of both Northern and Southern Resident killer whales revealed that salmonids (particularly Chinook) comprised 96% of the killer whale diet. However, most of these observations (>85%) were based on Northern residents; less information is available on the Southern Residents that routinely inhabit Puget Sound (Ford and Ellis 2006). The diet of transient killer whales is less well known, but is thought to be comprised primarily of harbor seals and to include other marine mammals such as sea lions, harbor porpoise, Dall's porpoise, minke whales and marine birds (Ford et al. 1998).

The movements and locations of SRKWs have been recorded by researchers, whale watchers and citizens since the early 1970s and a database of their distribution is maintained by The Whale Museum in Friday Harbor, Washington. Whales are most frequently observed in the San Juan Archipelago but are also found as far into Puget Sound as the southern portion of the South Sound (Figure 1)(Hauser et al. 2007, NMFS 2008). Southern Resident pods are present regularly in the Georgia Basin, and during warmer months all pods concentrate their activity from the south side of the San Juan Archipelago through Haro Strait northward to Boundary Pass (Hauser et al. 2007). Most transient sightings in the Puget Sound-Georgia Basin region are concentrated around southeastern Vancouver Island, the San Juan Archipelago, and the southern edge of the Gulf Islands. Transients appear to utilize a wider range of water depths and habitats than residents (NMFS).

Three main factors have been identified as potential threats to killer whales in Washington and British Columbia: reductions in prey availability, disturbance by underwater noise and vessel traffic, and exposure to environmental contaminants, particularly PCBs and PBDEs (NMFS 2008). Ford et al. (2010) suggests that declines in SRKW abundance in the mid 1990s were driven by a significant decline in range-wide abundance of Chinook salmon. NMFS has published a Final Recovery Plan that describes a recovery program designed to address each of the threats to the SRKW population. Due to their trophic position as apex predator, levels of contaminants such as polychlorinated biphenyl (PCBs) and dioxins in both resident and transient killer whales have been shown to be among the highest recorded (Ross et al. 2000, Krahn et al. 2007, Krahn et al. 2009).

Status

SRKW population: Photo-identification censuses of the SRKW population performed by the Center for Whale Research since the 1970s have shown several periods of growth and decline (Figure 2). Because the average life expectancy of killer whales is estimated to be 50 years and can extend to 80-90 years, the existing data on the SRKW populations have covered only a small portion of the lifespan. In response to a 20% population decline from 1996 to 2001, the SRKW stock was designated as depleted under the Marine Mammal Protection Act (MMPA) in 2003 and became listed as Endangered under the Endangered Species Act (ESA) in 2005. In 2006, NMFS designated approximately 2,500 square miles as critical habitat for Southern Residents. The designated area encompasses parts of Haro Strait, the waters around the San Juan Archipelago, the Strait of Juan de Fuca, and all of Puget Sound.

Transient population: Detailed estimates of population abundances for transient killer whale populations have not been made (NMFS 2008). It is hypothesized that historical transient killer whale populations experienced a large decline in abundance due to substantial prey losses in the early-to-mid 1900s (Springer et al. 2003). Because harbor seal populations in the region have increased over the last 30 years and currently are close to carrying capacity (Jeffries et al. 2003), it is believed that transients are no longer prey-limited (Ford et al. 2000). Approximately 225 transients have been identified in Washington, British Columbia, and southeastern Alaska (NMFS 2008) although current abundances are not known (NMFS 2008).

Trends

SRKW population: The historical population of Southern Residents in the mid- to late-1800s ws estimated to be approximately 200 whales (Krahn et al. 2002), although lack of data prior to the 1970s makes contributes to the uncertainty of this estimate. The capture of live killer whales for aquaria is thought to have removed approximately 50 Southern Resident and 5 transient killer whales between 1962 and 1977 (NMFS 2008). Since that time, the population has experienced fluctuations with periods of positive population growth followed by decline (Figure 2). Most notable was a substantial period of population growth between the mid 1980s and mid 1990s, during which total whale numbers expanded from 75 to nearly 98 animals. That period was followed by a brief period of decline, to 80 animals, followed by a moderate increase thereafter (Wiles 2004, Kriete 2007, NMFS 2008). The most recent estimate of 85 animals derives from a survey conducted in April 2009 (Center for Whale Research, (reported in PSP 2009)(Figure 2).

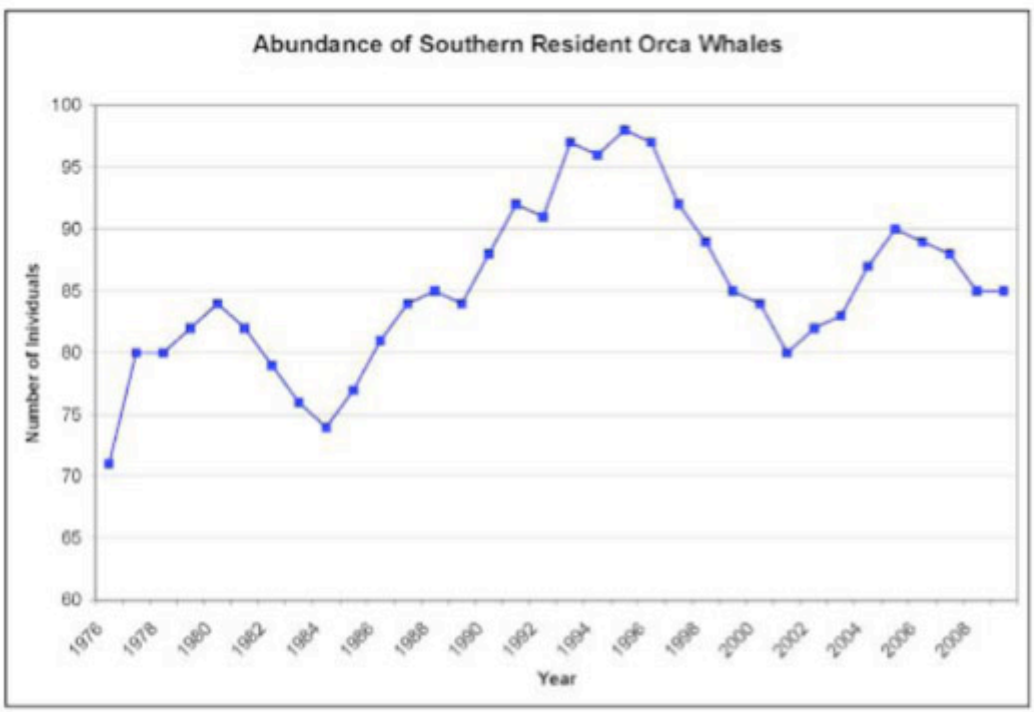


Figure 2. Abundance of Southern Resident killer whales from 1976-2009 (data from the Center for Whale Research)(reprinted from PSP 2009)

SRKW population predictions: Krahn et al. (2004) conducted a population viability analysis (PVA)(Morris and Doak 2002) to assess the future risk of extinction of the SRKW population, the predictions of which varied significantly according to the time period from which survival rates were estimated. Using the survival rates estimated from 1974-2003, they found that extinction probabilities for the SRKW whale populations ranged from <0.1-3% over the next 100 years and 2-42% over the next 300 years. However, extinction probabilities based on 1994-2003 survival rates ranged from 6-19% over the next 100 years and 68-94 % over the next 300 years (Krahn et al. 2004)

Transient population: Trends in abundance of the transient killer whale population cannot be estimated because accurate assessments of transient killer whale abundance have not been made.

Uncertainties

While the diets of Northern resident killer whales, which inhabit the coastal habitat of British Columbia and Alaska, have been well characterized (Ford and Ellis 2006), the extent to which diets of Northern resident killer whales are predictors of the diets of SRKW population (the primary users of Puget Sound habitats) remains under investigation. There is strong evidence for correlations between fluctuations in salmonids, especially Chinook salmon, and resident killer whales (Ford and Ellis 2006), but the drivers behind this relationship have not been elucidated. Furthermore, the unknown and potentially interactive effects of multiple stressors on killer whales introduces uncertainty in projections of future population abundances.

Summary

Killer whales are challenging to study because they spend much of their time below the water surface, are wide-ranging, and are highly migratory. Photo-identification and vigilant observations of predation events have allowed researchers to identify every individual in the SRKW population based on unique patterns and morphology, thereby facilitating accurate estimation of population abundance and diet of Resident killer whales. Human removal of SRKW appears to have driven population declines prior to the 1970s, yet 35 years after the removals for live capture ended, SRKW population numbers remain low. Data on transient killer whale populations are lacking.

Literature Cited

Ford, J. K. B., and G. M. Ellis. 2006. Selective foraging by fish-eating killer whales *Orcinus orca* in British Columbia. Marine Ecology-Progress Series 316:185-199.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 2000. Killer whales: The natural history and genealogy of *Orcinus orca* in British Columbia and Washington. UBC Press; University of Washington Press, Vancouver; Seattle.

Ford, J. K. B., G. M. Ellis, L. G. Barrett-Lennard, A. B. Morton, R. S. Palm, and K. C. Balcomb. 1998. Dietary specialization in two sympatric populations of killer whales (*Orcinus orca*) in coastal British Columbia and adjacent waters. Canadian Journal of Zoology-Revue Canadienne De Zoologie 76:1456-1471.

Ford, J. K. B., G. M. Ellis, P. F. Olesiuk, and K. C. Balcomb. 2010. Linking killer whale survival and prey abundance: food limitation in the oceans' apex predator? Biology Letters 6:139-142.

Hauser, D. D. W., M. G. Logsdon, E. E. Holmes, G. R. VanBlaricom, and R. W. Osborne. 2007. Summer distribution patterns of southern resident killer whales *Orcinus orca*: core areas and spatial segregation of social groups. Marine Ecology-Progress Series 351:301-310.

Jeffries, S., H. Huber, J. Calambokidis, and J. Laake. 2003. Trends and status of harbor seals in Washington state: 1978-1999. Journal of Wildlife Management 67:207-218.

Krahn, M. M., M. J. Ford, W. F. Perrin, P. R. Wade, R. P. Angliss, M. B. Hanson, B. L. Taylor, G. M. Ylitalo, M. E. Dahlheim, J. E. Stein, and R. S. Waples. 2004. 2004 Status review of Southern Resident killer whales (*Orcinus orca*) under the Endangered Species Act. Page 73 in N. T. Memorandum, editor.

Krahn, M. M., M. B. Hanson, R. W. Baird, D. G. Burrows, C. K. Emmons, J. K. B. Ford, L. L. Jones, D. P. Noren, P. S. Ross, G. S. Schorr, T. K. Collier, and R. H. Boyer. 2007. Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from Southern Resident killer whales. Marine Pollution Bulletin 54:1903-1911.

Krahn, M. M., M. B. Hanson, G. S. Schorr, C. K. Emmons, D. G. Burrows, J. L. Bolton, R. W. Baird, and G. M. Ylitalo. 2009. Effects of age, sex and reproductive status on persistent organic pollutant concentrations in "Southern Resident" killer whales. Marine Pollution Bulletin 58:1522-1529.

Krahn, M. M., P. R. Wade, S. T. Kalinowski, M. E. Dahlheim, B. L. Taylor, M. B. Hanson, G. M. Ylitalo, R. P. Angliss, J. E. Stein, and R. S. Waples. 2002. Status review of Southern Resident killer whales (*Orcinus orca*) under the Endangered Species Act. Page 133 in N. T. Memorandum, editor.

Kriete, B. 2007. Orcas in Puget Sound. Puget Sound Nearshore Partnership, Seattle.

Morris, W. F., and D. F. Doak. 2002. Quantitative conservation biology : theory and practice of population viability analysis. Sinauer Associates, Sunderland, Mass.

NMFS. 2008. Recovery plan for Southern Resident killer whales (*Orcinus orca*). National Marine Fisheries Service, Seattle.

PSP. 2009. Ecosystem Status & Trends. Puget Sound Partnership, Seattle, WA.

Ross, P. S., G. M. Ellis, M. G. Ikonomou, L. G. Barrett-Lennard, and R. F. Addison. 2000. High PCB Concentrations in Free-Ranging Pacific Killer Whales, *Orcinus orca*: Effects of Age, sex and dietary preference. Marine Pollution Bulletin 40:504-515.

Springer, A. M., J. A. Estes, G. B. van Vliet, T. M. Williams, D. F. Doak, E. M. Danner, K. A. Forney, and B. Pfister. 2003. Sequential megafaunal collapse in the North Pacific Ocean: An ongoing legacy of industrial whaling? Proceedings of the National Academy of Sciences of the United States of America 100:12223-12228.

Wiles, G. J. 2004. Washington State status report for the killer whale. Washington Department of Fish and Wildllife, Olympia, WA.

HABITATS

1. Eelgrass

Background

Eelgrass (*Zostera marina* L.) is an aquatic flowering plant common in tidelands and shallow waters along much of Puget Sound's shoreline. The species is restricted to soft-sediment habitats. Sexual reproduction occurs through seed production. Vegetative spread occurs via growth of below-ground rhizomes, which can result in the formation of large, dense beds. Eelgrass is widely recognized for its provision of important ecological functions (e.g., Hemminga and Duarte 2000, Duarte 2002), which in Puget Sound include the provision of energy to sustain diverse nearshore food webs (e.g., Simenstad and Wissmar 1985), as well as the creation of structurally complex habitat for a suite of species including herring, crab, shrimp, shellfish, waterfowl, and salmonids (Simenstad 1994, Heck et al. 2003, Mumford 2007). Eelgrass also stabilizes sediments and minimizes shoreline erosion (Duarte 2002). Because eelgrass requires growing conditions that include good water clarity and low nutrients, eelgrass abundance is considered to be an important indicator of estuarine health (e.g., Dennison et al. 1993, Hemminga and Duarte 2000). Industrial, agricultural and residential practices in upland areas and watersheds, and particularly activities that increase inputs of nutrients and suspended sediments, can negatively impact the growth of eelgrass. Direct physical impacts to eelgrass, such as propeller scour, overwater structures and shoreline development, also pose threats (Mumford 2007). In Washington, *Z. marina* has been designated a species of special concern by WDFW (WAC 220-110-250) and as critical habitat by the WDOE Shoreline Management Act (RCW 90.58).

Characteristics in Puget Sound

Eelgrass occurs in shallow soft sediments habitats throughout much of Puget Sound, with the notable exception of the southernmost portion (Mumford 2007, Gaeckle et al. 2009). Two habitat types are distinguishable based on nearshore geomorphology. Eelgrass flats are expansive, shallow beds typically located in bays, but also found at river deltas and shoals. Eelgrass fringe habitats consist of comparatively narrow, linear beds that follow the shoreline. In Puget Sound, eelgrass fringe habitats are more common than eelgrass flats, but because some flats are large in areal extent, the total area occupied by eelgrass is distributed roughly equally between the two habitat types. In the north Puget Sound and Saratoga-Whidbey regions, eelgrass occurs predominantly in large flats in Padilla and Samish Bays, which together account for approximately 25% of the total eelgrass in Puget Sound. By contrast, in the central, southern, and Strait of Juan de Fuca regions of Puget Sound, fringe beds are more common.

Multiple factors determine eelgrass distribution, including substrate availability, water clarity, wave energy, light attenuation, water temperature, tidal amplitude, and desiccation stress (Hemminga and Duarte 2000). In Puget Sound, the maximum depth to which eelgrass grows ranges from approximately 1.3 m below the low tide line (MLLW) to greater than 9 m deep. The deepest beds are found in the Strait of Juan de Fuca and the San Juan Islands (Gaeckle et al. 2009) .

WDNR Eelgrass Monitoring

The Nearshore Habitat Program of the Washington State Department of Natural Resources (WDNR) monitors eelgrass distribution and abundance through the Submerged Vegetation Monitoring Project (SVMP). The SVMP was established in 2000 to better understand eelgrass resources throughout Puget Sound and to detect temporal changes the distribution and abundance of eelgrass. The SVMP is part of the Puget Sound Assessment and Monitoring Program (PSAMP), a multi-agency effort coordinated by the Puget Sound Partnership to monitor diverse physical and biotic aspects of the Puget Sound ecosystem. Eelgrass is sampled annually at approximately 100 randomly-selected sites and 6 six permanent "core" sites (Figure 1). Sampling is performed at three spatial scales: Sound-wide, within regions, and within individual sites. Since monitoring began in 2000, more than 270 sites have been assessed. The SVMP was designed to detect changes that occur at annual and longer-term (5- and 10-year) temporal scales. The SVMP's primary programmatic performance measure is the ability to detect a 20% decline in eelgrass abundance over 10 years at the Sound-wide scale (Berry et al. 2003, Gaeckle et al. 2009). Data collection is carried out using underwater videography recorded along transects. Twelve to fifteen transects are sampled per site, oriented perpendicular to shore and sampled using a line-intercept method.

Figure 1. Distribution of site sampled in 2008 by SVMP sound -wide eelgrass monitoring study (reprinted from Gaeckle et al.2009 with permission from Nearshore Habitat Program, Washington Department of Natural Resources).

Status

Currently about 22,800 ± 4,500 hectares of eelgrass exist in greater Puget Sound, occupying approximately 43% of Puget Sound shoreline (Gaeckle et al. 2009). Eelgrass is more abundant in north Puget Sound than in the south. Approximately 91% of the estimated 9,859 ±2,603 hectares occurs in large, shallow embayments (Gaeckle et al. 2009). At individual sites, the areal extent of eelgrass ranges from less than 1 hectare to more than 3,000 hectares.

Trends

Trends in eelgrass distribution and abundance in Puget Sound prior to 2000 are difficult to establish due to a lack of long-term and broad-scale information preceding the initiation of the SVMP. Thom and Hallum (1990) performed a comprehensive examination of historical hydrographic charts, aerial photographs, WDFW survey information and other limited observations of eelgrass distribution in Puget Sound. The authors reported apparent declines in eelgrass abundance since the late 1800s in Bellingham Bay and the Snohomish River Delta, and an apparent increase in eelgrass abundance over approximately the same period in Padilla Bay.

Since monitoring began in 2000, the SVMP reports that the total area occupied by eelgrass in the Puget Sound has remained relatively stable (Gaeckle et al. 2009)(Figure 2). Despite this, site-level analyses suggest that in seven out of the last eight sampling periods, declines have been more frequent than increases (Gaeckle et al. 2009, Puget Sound Partnership 2009)(Table 1 and Figure 3), and sites with long-term declines outnumber sites with long-term increases (Gaeckle et al. 2009). Declines generally have occurred at smaller sites, while the extensive beds in the region, such as Padilla Bay and Samish Bay, remained stable. Gaeckle et al. (2009) conclude that the SVMP data suggests an overall pattern of slight declines in eelgrass throughout Puget Sound.

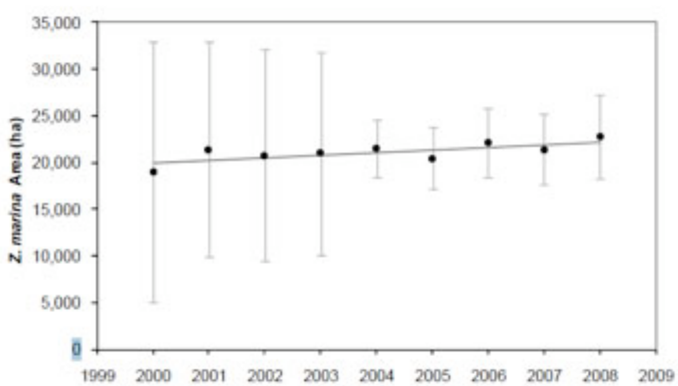


Figure 2. Sound-wide changes in area occupied by eelgrass from 2000 to 2008. Error bars represent 95% confidence intervals. The sharp improvement in precision in 2004 is due to increased sampling frequency at large sites. (reprinted from Gaeckle et al. 2009 with permission from Nearshore Habitat Program, Washington Department of Natural Resources)

Table 1. Results of a multiple parameter assessment of regional Z. marina condition based on data collected from 2000-2008. The number of measurable changes within a region was quantified and compared to the number of significant positive or negative changes (alpha = 0.05). CPS = Central Puget Sound, HDC = Hood Canal, NPS = North Puget Sound, SJS = San Juan/Straits, SWH = Saratoga/Whidbey. From Gaeckle et al. 2009, Nearshore Habitat Program, Washington Department of Natural Resources.

	CPS				HDC				NPS				SJS				SWH			
	No. Change Tests	Significant change	Positive change	Negative change	No. Change Tests	Significant change	Positive change	Negative change	No. Change Tests	Significant change	Positive change	Negative change	No. Change Tests	Significant change	Positive change	Negative change	No. Change Tests	Significant change	Positive change	Negative change
Site-level area	147	10	1	9	72	15	2	13	70	8	3	5	133	14	1	13	89	10	4	6
Deep edge depth	122	12	4	8	69	8	0	8	65	7	4	3	112	12	2	10	83	8	5	3
Shallow edge depth	122	16	7	9	68	13	5	8	64	8	4	4	111	12	6	6	83	15	7	8
5-year area trends	25	6	2	4	15	6	0	6	16	2	1	1	19	6	2	4	13	2	0	2
Proportion of significant results	0.11				0.19				0.12				0.12				0.14			
Proportion of significant positive results	0.32				0.16				0.48				0.25				0.43			
Proportion of significant negative results	**0.68**				**0.84**				**0.52**				**0.75**				**0.57**			

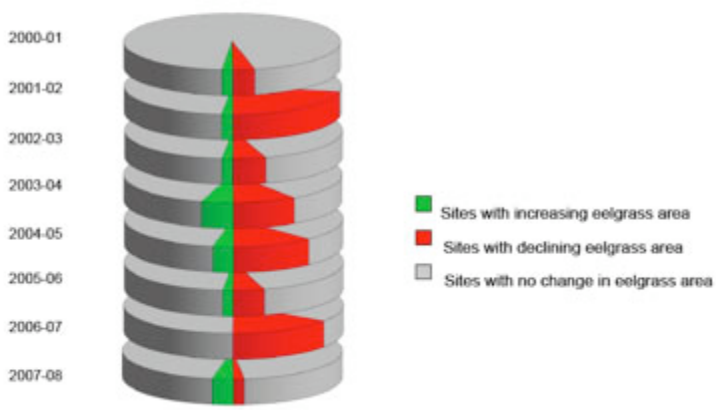


Figure 3. Eelgrass changes at individual sites. In seven of eight years of annual change, a greater proportion of sites showed statistically significant declines compared with increases in eelgrass area.(Nearshore Habitat Program, Washington Department of Natural Resources; reprinted from PSP 2009)

Uncertainties

The relative importance of the factors driving fluctuations in the distribution and abundance of eelgrass in Puget Sound is not well understood. Changes in key abiotic factors such as water clarity and nutrient levels may be important, yet analyses linking such abiotic data to eelgrass abundances have not been conducted. Consequently, the causes for declines in eelgrass cover documented by the SVMP are not known, nor are the ecological consequences of such declines for the taxa that utilize eelgrass habitat such as birds, invertebrates and fishes.

Summary

Eelgrass is critically important for maintaining nearshore ecosystem function and is recognized as a valuable indicator of ecosystem health. While the overall aerial extent of eelgrass in Puget Sound has shown no significant change over the past eight years, sharp local declines have been reported at some sites. The causes of these declines have not been established.

Literature Cited

Berry, H. D., A. T. Sewell, S. Wyllie-Echeverria, B. R. Reeves, T. F. Mumford, J. R. Skalski, R. C. Zimmerman, and J. Archer. 2003. Puget Sound Submerged Vegetation Monitoring Project: 2000-2002 Monitoring Report, Nearshore Habitat Program. Washington State Department of Resources, Olympia, WA.

Dennison, W. C., R. J. Orth, K. A. Moore, J. C. Stevenson, V. Carter, S. Kollar, P. W. Bergstrom, and R. A. Batiuk. 1993. Assessing water quality with submersed aquatic vegetation. BioScience 43:86-94.

Duarte, C. M. 2002. The future of seagrass meadows. Environmental Conservation 29:192-206.

Gaeckle, J. L., P.Dowty, H. Berry, and L. Ferrier. 2009. Puget Sound Submerged Vegetation Monitoring Project: 2008 Monitoring Report, Nearshore Habitat Program. Washington State Department of Natural Resources, Olympia, WA.

Heck, K. L., Jr., G. Hays, and R. J. Orth. 2003. Critical evaluation of the nursery role hypothesis for seagrass meadows. Marine Ecology Progress Series 253:123-136.

Hemminga, M. A., and C. M. Duarte. 2000. Seagrass Ecology. Cambridge University Press, Cambridge, UK; New York, NY.

Mumford, T. F. 2007. Kelp and eelgrass in Puget Sound. Seattle, WA.

Puget Sound Partnership. 2009. Ecosystem Status & Trends. Puget Sound Partnership.

Simenstad, C. 1994. Faunal associations and ecological interactions in seagrass communities of the Pacific Northwest coast. United States Environmental Protection Agency, Seattle, WA.

Simenstad, C. A., and R. C. Wissmar. 1985. Delta carbon-13 evidence of the origins and fates of organic carbon in estuarine and nearshore food webs. Marine Ecology Progress Series 22:141-152.

Thom, R. M., and L. Hallum. 1990. Long-term changes in the areal extent of tidal marshes, eelgrass meadows and kelp forests in Puget Sound, Wetland Ecosystem Team. United States Environmental Protection Agency

Kelp

Background

Kelps are large seaweeds in the order Laminariales that form dense canopies in temperate rocky intertidal and subtidal habitats less than 30 m in depth. The kelp flora of the Pacific Northwest is one of the most diverse in the world (Druehl 1969). Kelps are characterized by a highly dimorphic lifecycle consisting of a large diploid sporophytic (bed-forming) phase and a microscopic haploid gametophytic phase. In the Puget Sound region, bull kelp (*Nereocystis luetkeana*) occurs throughout Puget Sound and the Strait of Juan de Fuca, while the distribution of giant kelp (*Macrocystis integrifolia*) is restricted to the Strait of Fuca (Berry et al. 2005, Mumford 2007). Both form conspicuous floating canopies, or kelp beds. Sporophytes of *Nereocystis* are annual or semi-annual, whereas sporophytes of *Macrocystis* are perennial, persisting for several years. In addition to these dominant bed-forming taxa, numerous species of understory (non-floating) kelp occur subtidal habitats, many of which are present in southern and central Puget Sound (Mumford 2007).

Kelps are important primary producers. They contribute to Puget Sound food webs by providing food for herbivores and detritivores, and by releasing dissolved organic carbon (Duggins et al. 1989). In addition, kelps create important biogenic habitat that is utilized by fish, invertebrates, marine mammals, and birds (e.g., Ojeda and Santelices 1984, Graham 2004). Kelp can significantly alter the physical environment by modifying current and wave energy (Eckman et al. 1989) and this buffering capacity can influence the ecology of other organisms that utilize kelp environments for larval dispersal and settlement, for example rockfish (Carr 1991).

The extent and composition of kelp beds varies through time in response to natural and human-induced influences. In general, the distribution of kelp is determined by the amount of light available for photosynthesis, nutrient levels, grazers, physical disturbances, and toxic contaminants (reviewed in Mumford 2007). In addition to these external factors, demographic structure may play an important role in driving temporal dynamics of *Macrocystis* kelp beds through decreased fitness of older, more inbred populations (Raimondi et al. 2004, Reed et al. 2006).

Sea otters have been shown to be keystone predators in kelp forest ecosystem through their consumption of sea urchins, a major grazer of kelps (Estes and Palmisano 1974). In Washington state, otter populations have been slowly increasing since their reintroduction in 1969 and 1970 (Lance et al. 2004) following their extirpation through hunting in the 1900s. While they are more abundant on the open coast, otters have been observed as far east as Pillar Point in the Strait of Juan de Fuca (Lance et al. 2004, Laidre and Jameson 2006) where they have been shown to consumes a high proportion of urchins (Laidre and Jameson 2006). The potential for sea otters to expand into further into Puget Sound could affect kelp populations through trophic interactions. Furthermore, harvest of urchins by humans may be an important indirect driver of kelp populations in the Strait of Juan de Fuca; Berry et al. (2005) anecdotally observed that historic increases in urchin harvest rates were positively associated with increases in kelp abundances. However, in an experimental study, neither simulated fisheries removals nor simulated otter predation significantly affected the abundance of kelps in the San Juan Archipelago (Carter et al. 2007).

In addition to trophic interactions, climate changes associated with El Nino are known to cause short-term declines in kelp populations (e.g., Dayton and Tegner 1984), while the Pacific Decadal Oscillation could be driving changes over longer time periods. Substrate movement, as a result of altered nearshore hydrology and geomorphology, may also influence the amount of available habitat for attachment of kelps (Mumford 2007).

Due to their proximity to shore, kelps are likely to be subjected to anthropogenic impacts such as pollution discharge, nutrient influxes from urban and agricultural sources, increased turbidity, and increased rates of sedimentation (Dayton 1985, Mumford 2007). These can alter photosynthetic performance and growth of sporophytes and prevent settlement, growth, and reproduction of microscopic gametophytes. Toxic contaminants such as petroleum products are known to damage kelp by lowering photosynthetic and respiratory rates in meristematic tissue (Antrim et al. 1995).

Status

The Washington Department of Natural Resources (WDNR) conducts an annual inventory of canopy-forming kelp beds along the outer coast of Washington and the Strait of Juan de Fuca (approximately 360 km of shoreline). Inventories have been conducted annually since 1989 (with the exception of 1993) using aerial color-infrared photography (Van Wagenen 2004). In 2005, Berry et al. (2005) reported a total of approximately 1,700 hectares of floating kelp (Nereocystis and Macrocystis) on Washington's outer coast and the Strait of Juan de Fuca.

Trends

Prior to the initiation of annual floating kelp inventories by WDNR, Thom and Hallum (1990) reviewed several sources of historical data and found evidence that floating kelp had increased by 58 percent since the first European mapping in the 1850s. The largest increases were observed in the most populated areas such as central and south Puget Sound, but anecdotal evidence for losses in central Puget Sound were also noted. Between 1989 and 2004, the annual inventories conducted by WDNR for floating kelp at 66 shoreline sections on the outer coast and in the Strait of Juan de Fuca show high year-to-year variation, ranging from 722 hectares in 1997 to 2,575 hectares in 2000 (Berry et al. 2005)(Figure 1). Between the two species of floating kelp, M. integrifolia canopy area was more stable over time than N. luetkeana canopy, potentially due to their differing life histories. From 1989 to 2004, total floating kelp canopy area increased significantly ($p<0.01$), but these increases were restricted to the Outer Coast and the Western Strait of Juan de Fuca; kelps in the Eastern Juan de Fuca region showed no trend (Berry et al. 2005)(Figure 2). At the smallest scale (5-15km of shoreline), kelp area increased significantly in 18 sections, decreased significantly in 1 section, and did not change significantly in 47 sections (Berry et al. 2005)(Figure 3). A significant decrease in kelp canopy area was detected near Protection Island. Kelp canopies in this area have declined gradually from more than 10 hectares in 1989 and 1990 to less than 1 hectare annually since 1994 (Berry et al. 2005)(Figure 3).

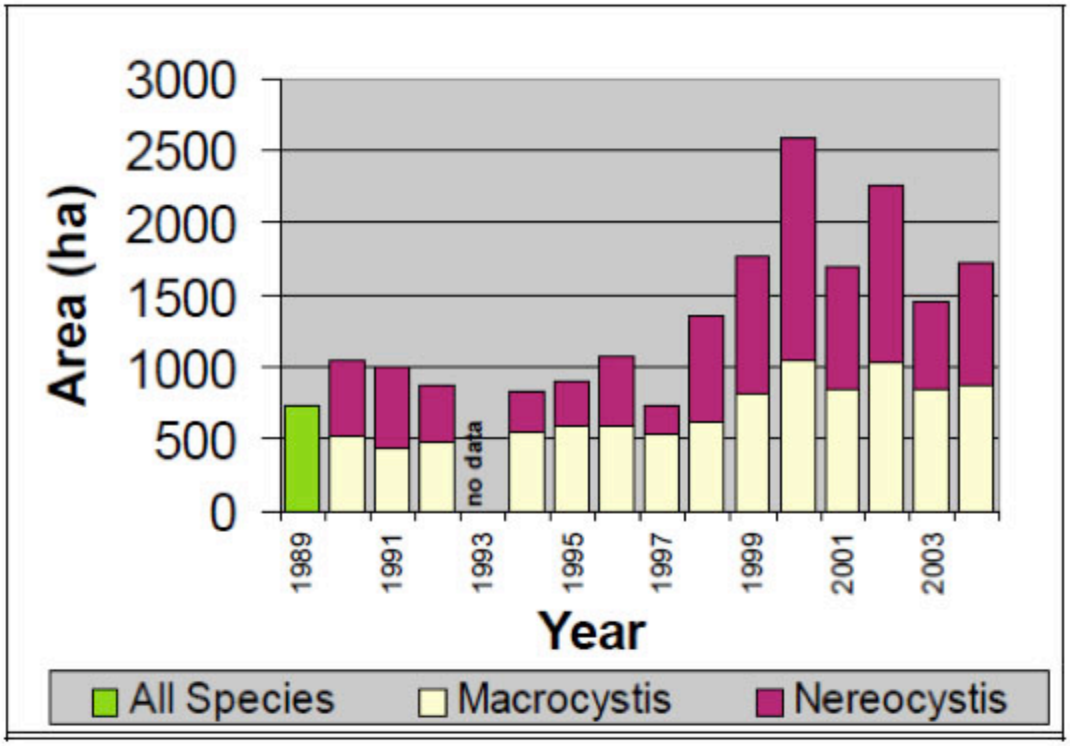


Figure 1. Floating Kelp Canopy Area on Washington's outer coast and the Strait of Juan de Fuca, 1989-2004 (reprinted from Berry et al. 2005 with permission from Nearshore Habitat Program, Washington Department of Natural Resources).

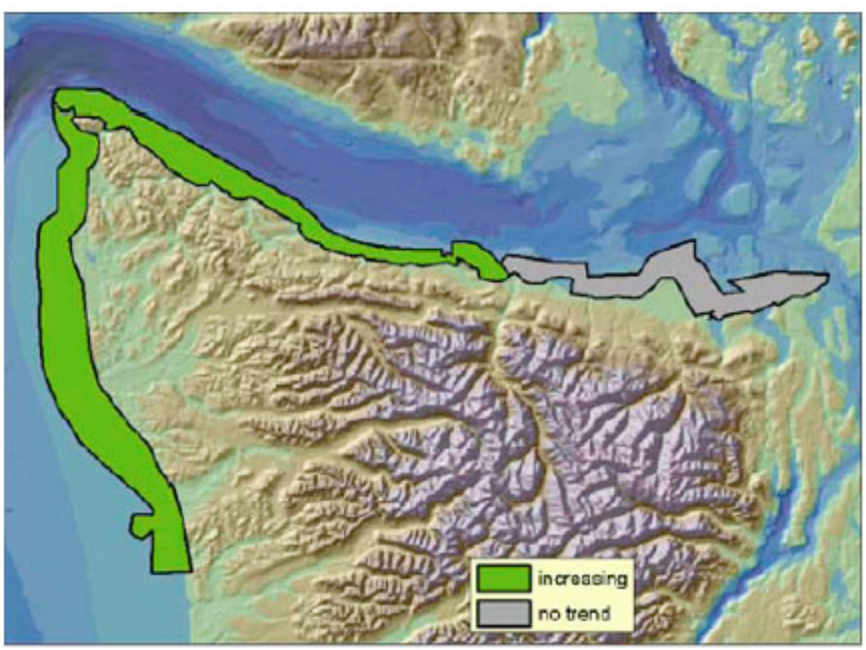


Figure 2. Region changes in kelp canopy area ($p < 0.01$), based on annual surveys between 1989 and 2004 (reprinted from Berry et al. 2005 with permission from Nearshore Habitat Program, Washington Department of Natural Resources).

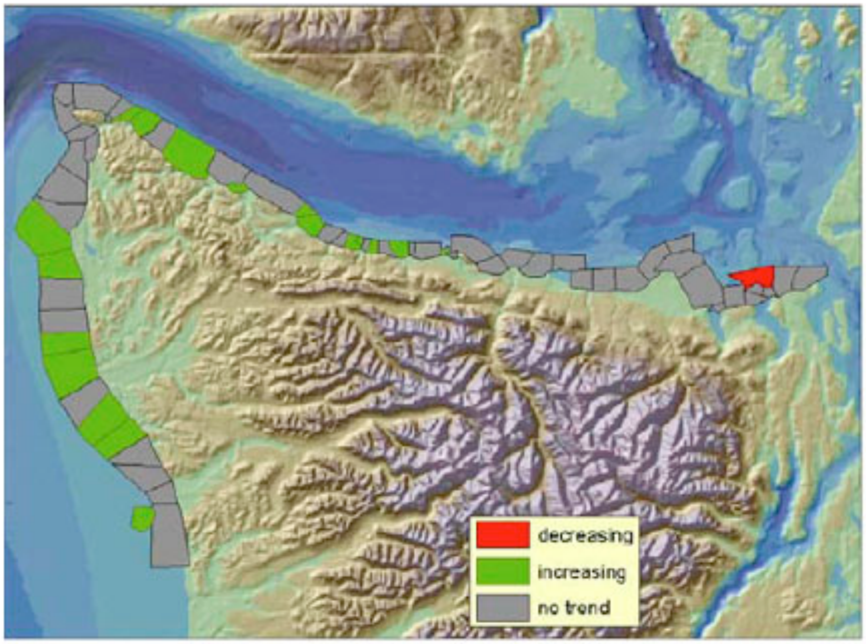


Figure 3. Shoreline sections with significant changes in kelp canopy area ($p < 0.01$), based on annual surveys between 1989 and 2004 (reprinted from Berry et al. 2005 with permission from Nearshore Habitat Program, Washington Department of Natural Resources).

Despite these findings, Mumford (2007) notes multiple anecdotal accounts of kelp bed losses around Marrowstone, Bainbridge, and Fox islands as well as personal observations of the loss of small kelp beds in southern Puget Sound at Itsami Ledge, Devils Head and Dickenson Point. A large Nereocystis bed on Dallas Bank, north of Protection Island in the Strait of Juan de Fuca, has almost totally disappeared since 1989 (Mumford 2007). Other anecdotal observations indicate substantial declines in bull kelp abundance in the San Juan Archipelago and the Strait of Georgia. Taken together, the observations could suggest widespread declines in bull kelp in Puget Sound. The causes of these changes are not known.

Uncertainties

The long-term WDNR dataset provides important insight into how the aerial extent of kelp canopies has changed over time, yet there may be potential biases associated with this method. Berry et al. (2005) notes that observed trends could be subject to methodological artifacts related to environmental factors (primarily tidal height and current speed) that introduce uncertainty or bias in the monitoring data. Both tides and currents have been shown to affect apparent *Nereocystis* canopy area as observed by photographs taken from the adjacent shoreline (Britton-Simmons et al. 2008). Consequently, it is possible that some of the observed variation in kelp canopy cover may be inflated by changes in the conditions under which the photographs were taken.

The WDNR monitoring programs focuses on the two species of floating kelp (*Nereocystis* and *Macrocystis*) native to the region. However, understory (non-floating) kelps are abundant and widely distributed throughout Puget Sound, where their ecological importance could equal that of the canopy-forming kelps. Effective monitoring of subtidal kelp populations is not yet possible, although use of towed video arrays holds promise (Mumford 2007). Furthermore, little is known about the ecology of the microscopic gametophyte phase of kelps due to the difficulty of studying them in situ (Mumford 2007). Failure in settlement, growth, or reproduction in microscopic stages will result in disappearance of the conspicuous sporophytic phases.

Summary

Kelps are important primary producers and create important biogenic habitat in Puget Sound ecosystems. Annual aerial surveys of floating kelp canopies conducted by WDNR show that between 1898 and 2004 floating canopies increased in outer coastal areas an in the western Strait of Juan de Fuca. Floating kelp canopies in the eastern Strait of Juan de Fuca showed no statistical change over the same period. Anecdotal evidence indicates sharp local declines in kelp abundance in southern and central Puget Sound and the San Juan Archipelago and calls for new investigations and expansion of kelp surveys.

Literature Cited

Antrim, L. D., R. D. Thom, W. W. Gardiner, V. W. Cullinan, D. K. Shreffler, and R. W. Bienert. 1995. Effects of petroleum products on bull kelp (*Nereocystis luetkeana*). Marine Biology 122:23-31.

Berry, H. D., T. F. Mumford, and P. Dowty. 2005. Using historical data to estimate changes in floating kelp (*Nereocystis luetkeana* and *Macrocystis integrifolia*) in Puget Sound, Washington. Puget Sound Georgia Basin Research Conference. Nearshore Habitat Program, Washington Department of Natural Resources, Olympia, WA.

Britton-Simmons, K., J. E. Eckman, and D. O. Duggins. 2008. Effect of tidal currents and tidal stage on estimates of bed size in the kelp *Nereocystis luetkeana*. Marine Ecology Progress Series. 355:95.

Carr, M. H. 1991. Habitat selection and recruitment of an assemblage of temperate zone reef fishes. Journal of Experimental Marine Biology and Ecology 146:113-137.

Carter, S. K., G. R. VanBlaricom, and B. L. Allen. 2007. Testing the generality of the trophic cascade paradigm for sea otters: a case study with kelp forests in northern Washington, USA. Hydrobiologia 579:233-249.

Dayton, P. K. 1985. Ecology of Kelp Communities. Annual Review of Ecology and Systematics 16:215-245.

Dayton, P. K., and M. J. Tegner. 1984. Catastrophic storms, El Nino, and patch stability in a southern California kelp community. Science 224:283-285.

Druehl, L. D. 1969. The northeast Pacific rim distribution of Laminariales. Pages 161-170 in International Seaweed Symposium.

Duggins, D. O., C. A. Simenstad, and J. A. Estes. 1989. Magnification of secondary production by kelp detritus in coastal marine ecosystems. Science 245:170-173.

Eckman, J. E., D. O. Duggins, and A. T. Sewell. 1989. Ecology of under story kelp environments. I. Effects of kelps on flow and particle transport near the bottom. Journal of Experimental Marine Biology and Ecology 129:173-187.

Estes, J. A., and J. F. Palmisano. 1974. Sea Otters: Their role in structuring nearshore communities. Science 185:1058-1060.

Graham, M. H. 2004. Effects of Local Deforestation on the Diversity and Structure of Southern California Giant Kelp Forest Food Webs. Ecosystems 7:341-357.

Laidre, K. L., and R. J. Jameson. 2006. Foraging Patterns and Prey Selection in an Increasing and Expanding Sea Otter Population. Journal of Mammalogy 87:799-807.

Lance, M. M., S. A. Richardson, and H. L. Allen. 2004. Washington State Recovery Plan for the Sea Otter. Washington Dept. of Fish and Wildlife, Wildlife Program, Olympia, WA.

Mumford, T. F. 2007. Kelp and eelgrass in Puget Sound. Seattle District, U.S. Army Corps of Engineers, Puget Sound Nearshore Partnership, Seattle, WA.

Ojeda, F. P., and B. Santelices. 1984. Invertebrate communities in holdfasts of kelp *Macrocystis pyrifera* from Southern Chile. Marine Ecology Progress Series 16:65-73.

Raimondi, P. T., D. C. Reed, B. Gaylord, and L. Washburn. 2004. Effects of self-fertilization in the giant kelp, *Macrocystis pyrifera*. Ecology 85:3267-3276.

Reed, D. C., P. T. Raimondi, L. Washburn, B. Gaylord, B. P. Kinlan, and P. T. Drake. 2006. A metapopulation perspective on patch dynamics and connectivity in giant kelp.in J. P. Kritzer and P. F. Sale, editors. Marine Metapopulations. Elsevier Academic Press, Amsterdam; Boston.

Thom, R. M., and L. Hallum. 1990. Long-term changes in the areal extent of tidal marshes, eelgrass meadows and kelp forests in Puget Sound, Wetland Ecosystem Team. United States Environmental Protection Agency.

Van Wagenen, R. 2004. Washington coastal kelp resources, Summer 2004. Nearshore Habitat Program, Washington Department of Natural Resources, Olympia, WA.

Tidal Wetlands

Background

Tidal wetlands are highly productive ecosystems that provide a variety of resources and ecosystem functions to Puget Sound biota and humans. In this report, tidal wetlands refer to both estuarine (intertidal) and riverine tidal (tidally-influenced freshwater) wetlands along the Puget Sound shoreline. Wetlands provide important ecosystem roles, directly regulating hydrologic and biogeochemical processes and supporing high rates of biological productivity (Mitsch and Gosselink 2007). They also are a key habitat for a suite of fish, amphibian, invertebrate and bird species including chum and Chinook salmon, herring, Dungeness crabs and Great Blue Herons (e.g., McMillan et al. 1995, Simenstad and Cordell 2000, Eissinger 2007 , Stick and Lindquist 2009). Because of the fjord-like topography in Puget Sound, tidal wetlands are predominantly associated with the major rivers. The steep, rocky bathymetry and topography limit the existence of extensive intertidal areas or the deposition of sediments on which vegetated wetland might occur (Boule 1981). Low gradient rivers combined with substantial tidal ranges create large intertidal areas in river floodplains that contain plant communities strongly controlled by a substantial amount of freshwater runoff. Tidal wetlands in Puget Sound have experienced significant losses and degradation as a result of development and other land uses.

Status

Collins and Sheikh (2005) characterized tidal wetland habitat across the sub-basins of Puget Sound (Figure 1) using both aerial and oblique photographs taken from 1998 – 2000 as part of a detailed study comparing the extent and nature of current and historical wetlands. They found that nearly half of the current tidal marsh area is in located in the Skagit, Stillaguamish and Samish river deltas and that the median size of a tidal wetland complex is 0.57 hectares (Figure 2)(Collins and Sheikh 2005). They estimate that there are currently 5,650 hectares of tidal wetland habitat in Puget Sound (Collins and Sheikh 2005). In a more detailed analysis of the composition of wetlands in river deltas, they found that the dominant type of tidal wetland in the river deltas of Puget Sound is currently estuarine-emergent wetland relative to the less frequent estuarine scrub-shrub and riverine habitat types (Figure 3)(Collins and Sheikh 2005).

Figure 1. Sub-basins of Puget Sound as defined by Collins and Sheikh (2005). Reprinted with permission from Collins and Sheikh (2005).

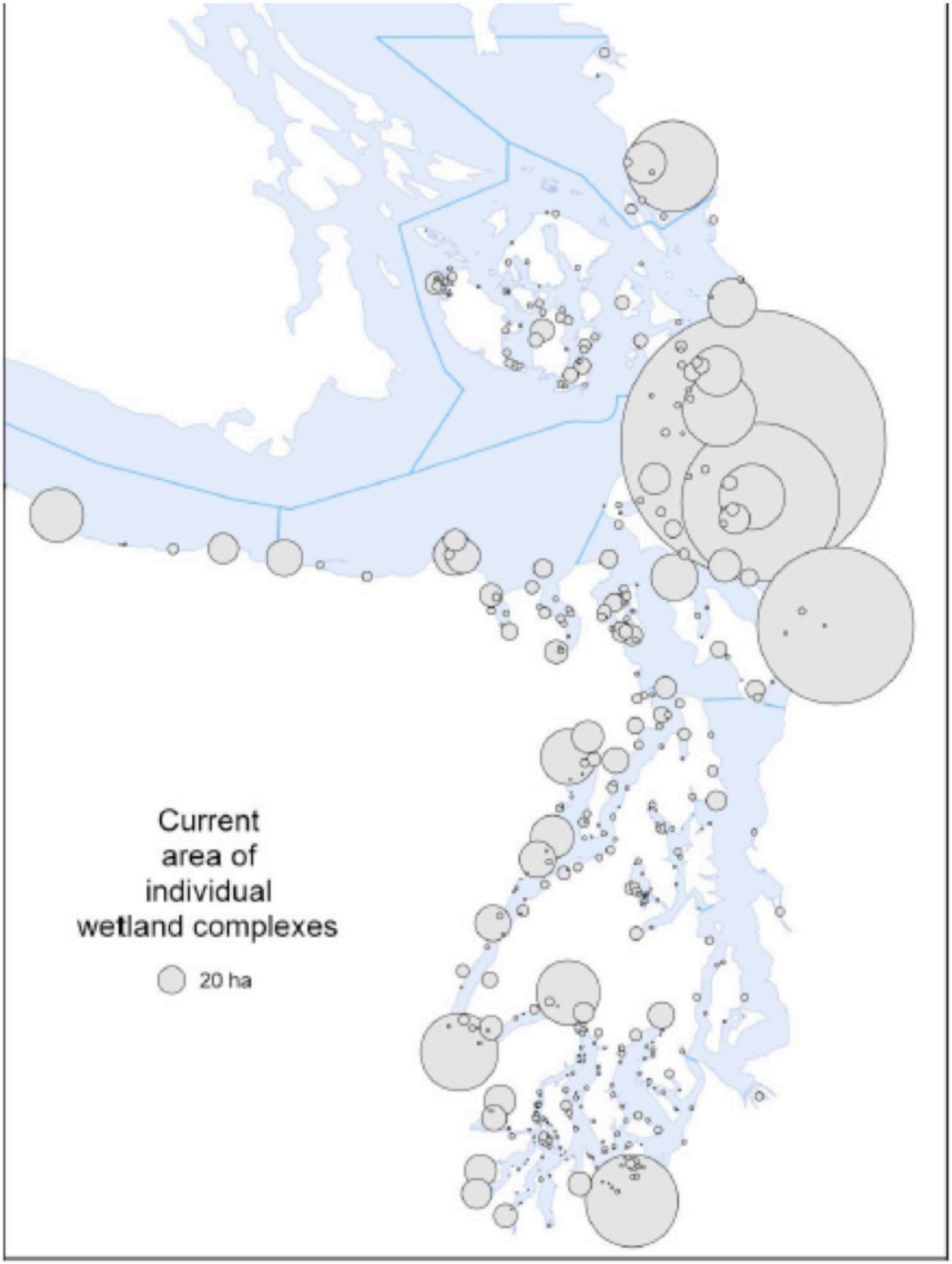

Current
area of
individual
wetland complexes
20 ha

Figure 2. Current area of individual wetland complexes (note: in all pie diagrams, wetland is proportional to the symbol area (reprinted with permission from Collins and Sheikh 2005)

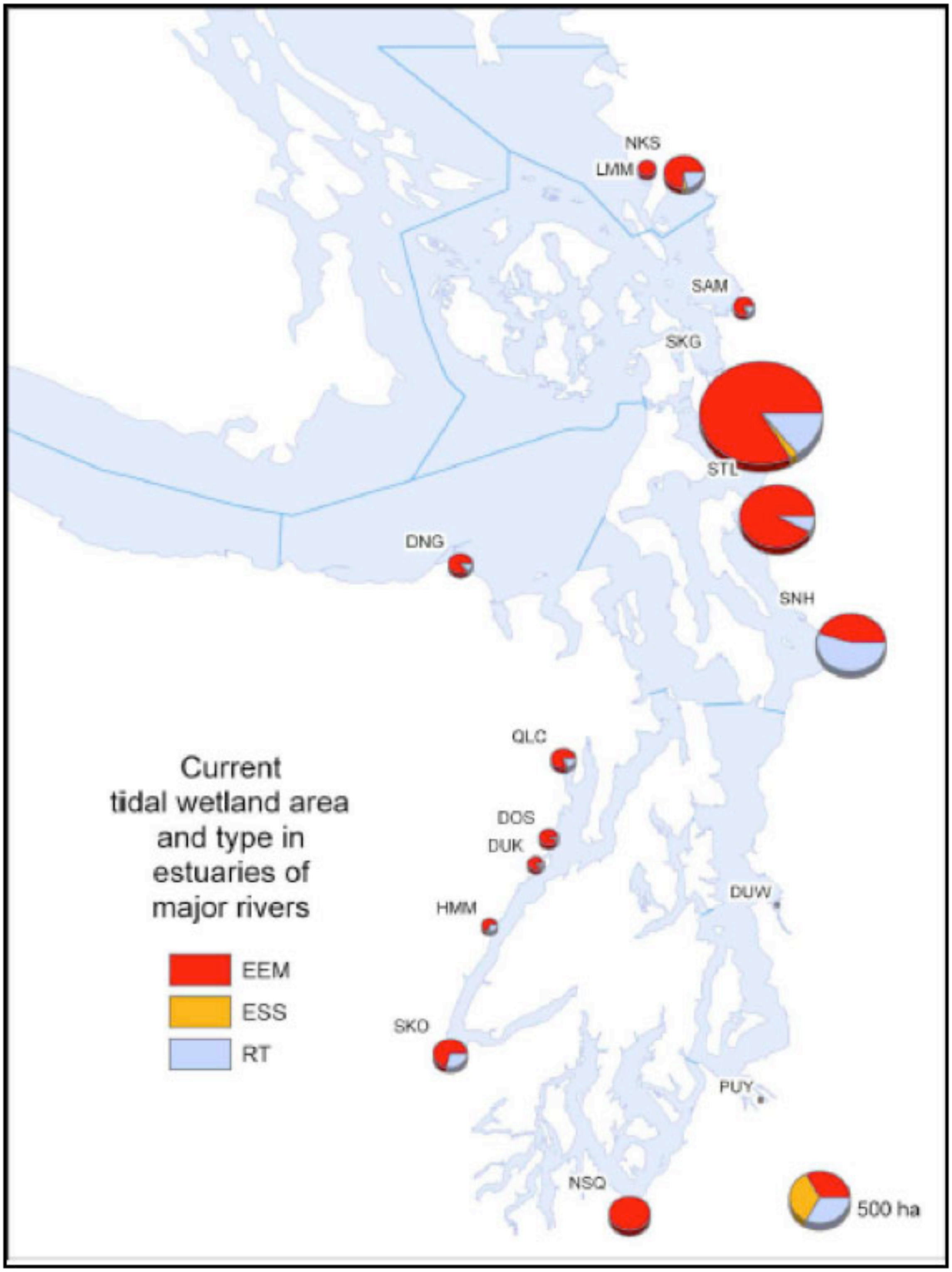


Figure 3. Relative area of current tidal wetland types in the estuaries of major rivers draining the Cascade Range and Olympic Mountains. EEM- estuarine emergent wetland; ESS- estuarine scrub-shrub wetland; RT- riverine-tidal wetland. (note: in all pie diagrams, wetland is proportional to the symbol area) (reprinted with permission from Collins and Sheikh 2005)

Trends

Several quantitative investigations into the degree of alteration of tidal wetlands have been conducted in Puget Sound. The earliest and most comprehensive assessment of areal coverage of tidal wetlands occurred in the mid 1880s by a Snohomish resident for the purposes of assessing agricultural development potential (Nesbit 1885). This endeavor used navigation maps, interviews with residents, and field observations to document the extent of tidal marshes and swamps (inclusive of saltmarsh and freshwater marsh) throughout Washington State from ca.1883. It found that tidal marshes greatly exceeded tidal flats in area on Puget Sound and that freshwater marshes were three to four times as great in extent as compared to the tidal marshes. Based on this early surveying effort by Nesbit (1885) , Thom and Hallum (1990) estimated approximately 26,792 hectares of tidal wetlands in seven of the nine counties bordering Puget Sound in the late 1800s. As such, approximately 38% of tidal marshes in Puget Sound may have already been converted to agricultural and urban land uses by the late 1800s (Nesbit 1885, Collins and Sheikh 2005).

The historic extent of tidal wetlands in Puget Sound was also recorded on topographic charts known as "T-sheets," which were produced by the U.S. Coast Survey and the U.S. Coast and Geodetic Survey in 1884-1908. A review of comparisons between the T-sheets and more current sources including U.S. Geologic Survey topographic maps (produced in the 1970s) was conducted by Thom and Hallum (1990). This effort also drew upon analyses by Bortelson et al. (1980) and Boule et al. (1983). This investigation revealed that the most substantial intertidal wetland losses occurred in the Snohomish, Duwamish and Puyallup river deltas, reported to have experienced loss of 32 %, 100%, and 99% respectively. Subaerial wetland loss (defined as those wetlands landward of the general saltwater shoreline, but exclusive of intertidal wetlands) was also significant, with total losses of approximately 73% in river deltas throughout Puget Sound since the late 1800s (Bortleson et al. 1980, Thom and Hallum 1990).

More recently, Collins et al. (2003) reconstructed historical environments of several estuaries in northern Puget Sound and concluded that a considerable amount of tidal wetland had already been converted to agricultural and other land uses prior to development of the T-sheets, particularly estuarine scrub-shrub and riverine tidal environments, which were the basis of previous studies. To provide a comprehensive assessment, the Washington Department of Natural Resources (WDNR) collaborated with the University of Washington (UW) to characterize the historic and current distribution, type, and amount of tidal wetlands in Puget Sound (2005). Collins and Sheikh (2005) used a number of other sources that supplemented and cross-referenced the T-sheets, including records of federal land survey, aerial photographs, the survey conducted by Nesbit (1885) and soil surveys. They developed an atlas of pre-settlement (mid 1880s) riverine and nearshore habitats consisting of a spatially explicit digital database based on a landform and process-based classification of nearshore wetlands (see Collins and Sheikh (2005) for a complete summary of methods and results). They estimated the historic area of wetland habitat in Puget Sound to be 29,500 acres, indicating that the current tidal wetlands are 17 – 19% of their historical extent (Collins and Sheikh 2005). They found that the Whidbey basin (which includes the Snohomish, Skagit and Stillaguamish rivers) has experienced the largest total loss of areal coverage followed by the Sand Juan Islands/North Coast (which includes the Padilla Bay part of the greater Skagit River delta, and the Samish River), the Fraser Lowland (which includes the Lummi and Nooksack rivers), and the Central Sound (which includes the Duwamish and Puyallup rivers) (Figure 4). Moreover, the median size of individual wetlands has decreased over time from approximately 0.93 hectares to 0.57 hectares (Figures 2 and 5)(Collins and Sheikh 2005). The composition of river delta wetlands has also undergone a major shift such that the relative abundance of emergent scrub-shrub and riverine-tidal vegetation were historically higher than current levels (Figures 3 and 6)(Collins and Sheikh 2005).

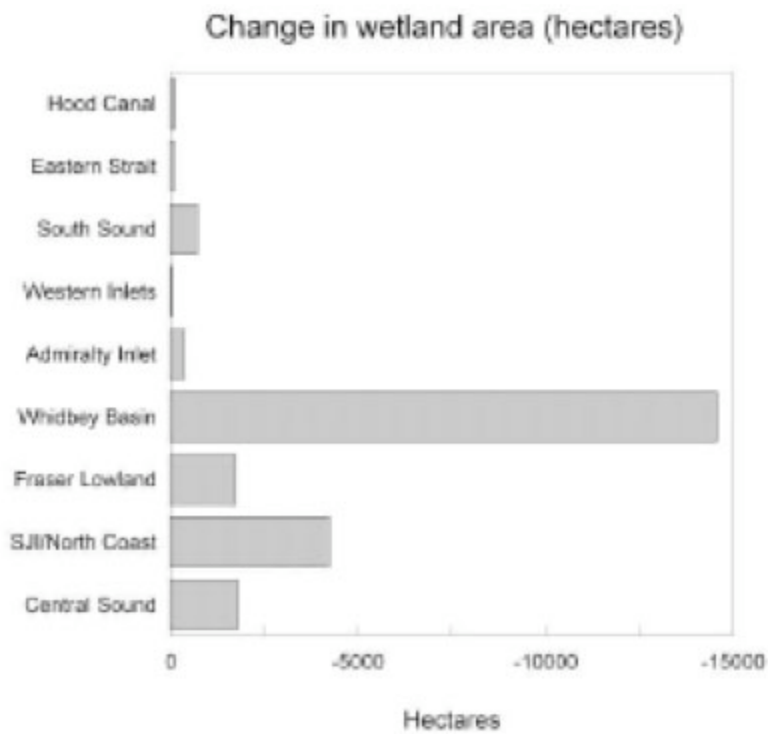


Figure 4. Change in wetland area (hectares) in Puget Sound (reprinted with permission from Collins and Sheikh 2005)

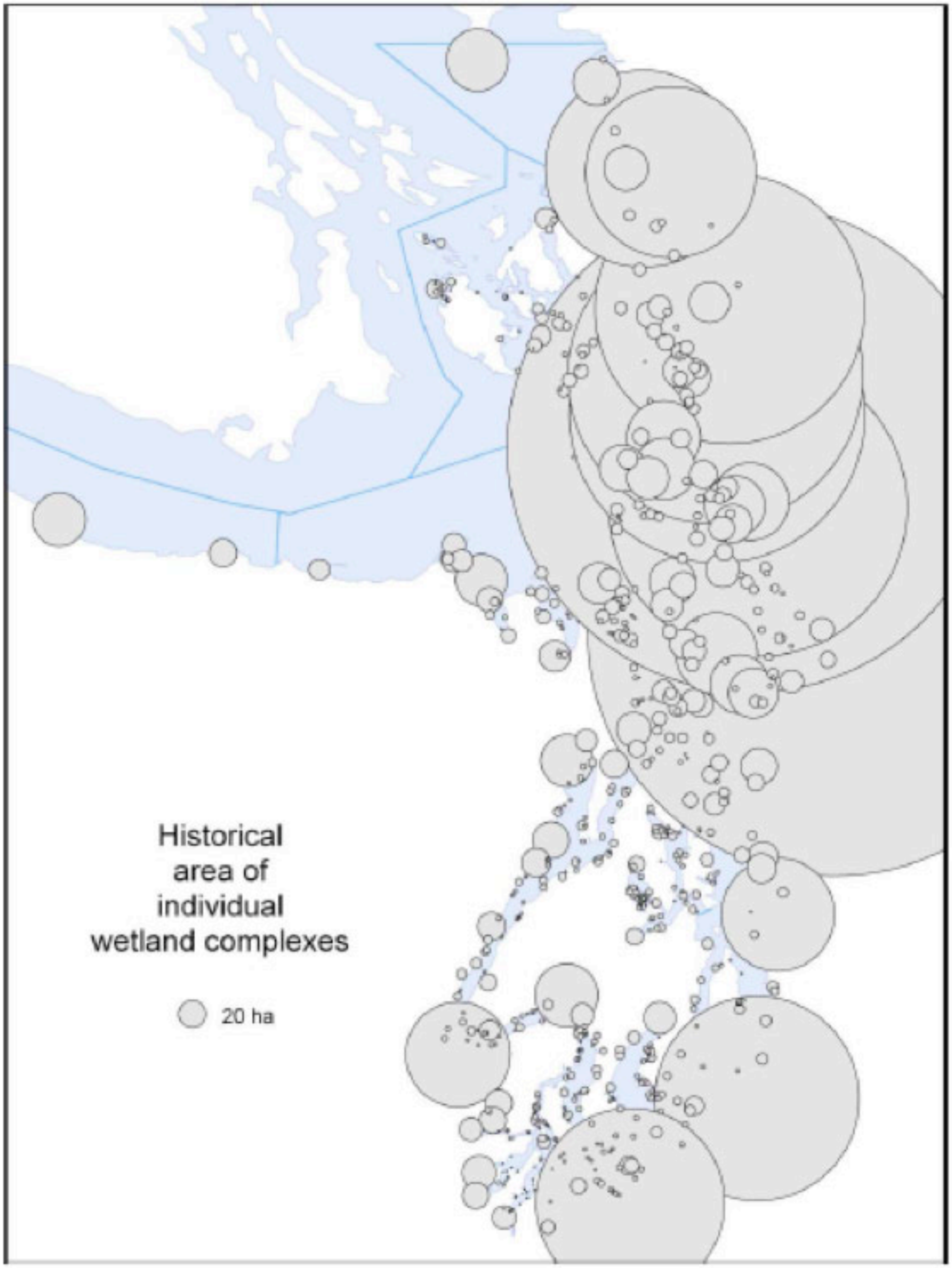


Figure 5. Historical area of individual wetland complexes (note: in all pie diagrams, wetland is proportional to the symbol area (reprinted with permission from Collins and Sheikh 2005)

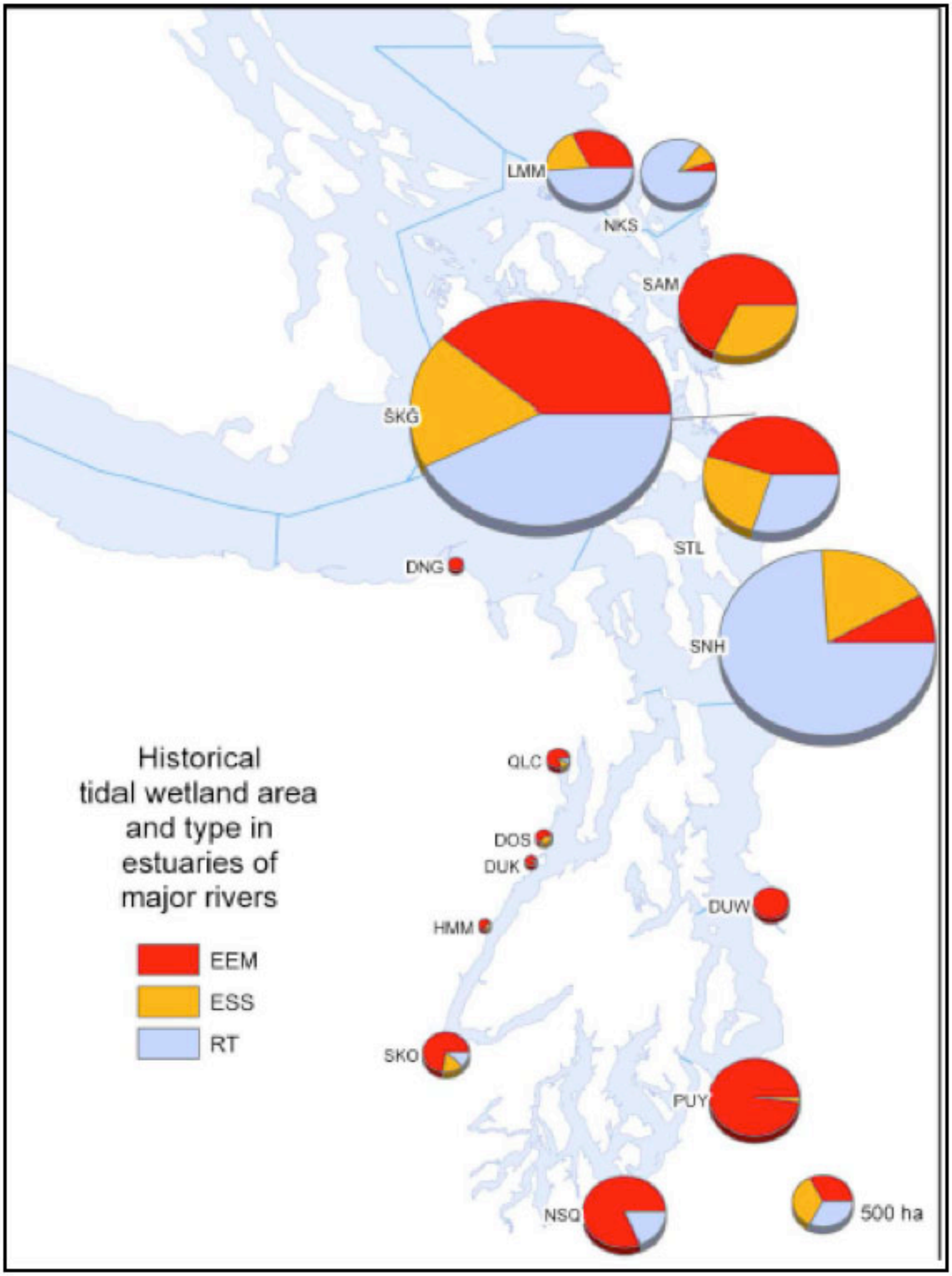


Figure 6. Relative area of historical tidal wetland types in the estuaries of major rivers draining the Cascade Range and Olympic Mountains. EEM- estuarine emergent wetland; ESS- estuarine

scrub-shrub wetland; RT- riverine-tidal wetland. (note: in all pie diagrams, wetland is proportional to the symbol area) (reprinted with permission from Collins and Sheikh 2005)

Uncertainties

Assessing the degree to which wetlands have changed over time is challenging. As with any analysis of historical trends, the frame of reference (baseline) can dictate the perception of change (e.g., Jackson et al. 2001), yet historical accounts are often less quantitative and thereby more subjective (Thom and Hallum 1990, Collins and Sheikh 2005). The use of historic maps from different sources is hindered by differences in terminology with respect to classifications of wetland hydrology, habitat or vegetation. Despite these challenges, the current efforts to recreate a quantitative picture of the extent and nature of historic wetlands have taken substantial measures to account for these difficulties (Thom and Hallum 1990, Collins and Sheikh 2005). The similarity of independent estimates derived from disparate sources strengthens confidence in them. Both Thom Hallum (1990) and Collins and Sheikh (2005) acknowledge that their estimates of historic wetland area may still be lower than their true extent given the limitations in the available data. The Puget Sound Nearshore Ecosystem Restoration Project (PSNERP) is currently conducting a closer investigation of intertidal wetlands using the database created by WDNR and UW. This effort is ongoing and will likely yield a more detailed analysis of wetland change in Puget Sound. While there has been much recent and ongoing efforts to restore wetlands in Puget Sound, the effectiveness and long-term sustainability has not been determined for the entire Puget Sound, though monitoring programs are used to document progress towards this end. The existing comparisons between current and historic wetlands do not currently separate restored wetlands from natural ones.

Summary

Tidal wetlands play an integral role in the hydrology, chemistry and nearshore ecosystem of Puget Sound and have experienced significant declines as a result of industrial uses, agricultural uses, and other types of human development. While much of the wetland loss and alteration occurred after 1900, dredging and channeling of large river deltas began as early as the 1850s. There have been several investigations into wetland change since pre-industrial times, each utilizing divergent or common data sources and deriving generally consistent estimates. The most recent and comprehensive assessment documents that the current area of tidal wetlands in Puget Sound is 17-19 % of historic levels and that most of the loss has occurred in the Whidbey Basin (Collins and Sheikh 2005). Ongoing investigations by PSNERP stand to shed more light on the extent and nature of current and historic wetland alterations in Puget Sound. Currently, efforts to restore estuarine and tidal wetlands hold promise for recovering lost ecosystem function.

Literature Cited

Bortleson, G. C., M. Chrzastowski, and A. K. Helgerson. 1980. Historical changes of shoreline and wetland at eleven major deltas in the Puget Sound region, Washington. U.S. Geological Survey, U. S. Dept. of, Justice, U.S. Bureau of Indian, Affairs, Reston, Va.

Boule, M. 1981. Tidal wetlands of the Puget Sound region, Washington. Wetlands 1:47-60.

Boule, M. E., N. Olmsted, and T. Miller. 1983. Inventory of wetland resources and evaluation of wetland management in western Washington. The Dept. of Ecology, Olympia, WA.

Collins, B. D., D. R. Montgomery, and A. J. Sheikh. 2003. Reconstructing the historical riverine landscape of the Puget Lowland Pages 79-128 in D. R. Montgomery, S. M. Bolton, D. B. Booth, and L. Wall, editors. Restoration of Puget Sound Rivers. University of Washington Press, Seattle, WA.

Collins, B. D., and A. J. Sheikh. 2005. Historical reconstruction, classification, and change analysis of Puget Sound tidal marshes. University of Washington (Seattle, WA) and the Nearshore Habitat Program, Washington State Dept. of Natural Resources, Olympia, WA.

Eissinger, A. M. 2007 Great Blue Herons in Puget Sound. . Seattle, WA.

Jackson, J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, J. Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner, and R. R. Warner. 2001. Historical Overfishing and the Recent Collapse of Coastal Ecosystems. Science 293:629-637.

McMillan, R. O., D. A. Armstrong, and P. A. Dinnel. 1995. Comparison of intertidal habitat use and growth rates of two northern Puget Sound cohorts of 0+ age Dungeness crab, Cancer magister Estuaries 18:390-398.

Mitsch, W. J., and J. G. Gosselink. 2007. Wetlands. 4th edition. John Wiley & Sons, Inc., Hoboken, NJ.

Nesbit, D. M. 1885. Tide marshes of the United States. U.S. Dept. of Agriculture, Washington, D.C.

Simenstad, C. A., and J. R. Cordell. 2000. Ecological assessment criteria for restoring anadromous salmonid habitat in Pacific Northwest estuaries. Ecological Engineering 15:283-302.

Stick, K. C., and A. Lindquist. 2009. 2008 Washington State Herring Stock Status Report. Stock Status Report No. FPA 09-05, Olympia, WA.

Thom, R. M., and L. Hallum. 1990. Long-term changes in the areal extent of tidal marshes, eelgrass meadows and kelp forests of Puget Sound. Wetland Ecosystem Team, Fisheries Research Institute, School of Fisheries, University of Washington, Seattle, WA.

Water Quality

Puget Sound is unique in the lower 48 Unites States because of its fjord-like physiography, inland extent, wide range of depths, and urbanized watersheds and shorelines. Limited exchange of seawater between sub-basins within Puget Sound can result in long residence times, potentially increasing the susceptibility of biota to contamination introduced through human activities. The varied habitats within Puget Sound support multiple life history stages of many species, potentially exposing sensitive life stages to contamination. There are multiple water quality concerns in Puget Sound:

- Levels of toxic contaminants in biota that live or feed in Puget Sound.
- The eutrophication of marine waters, producing hypoxic and anoxic regions.
- Wastewater contamination, principally from combined sewer overflows or septic systems
- Harmful algal blooms, which introduce toxins that enter the food web
- Acidification of marine waters, and the adverse ecological effects that result.

Degradation of water quality in Puget Sound occurs through three primary mechanisms. The first is through the introduction of toxic contaminants, primarily comprising manufactured synthetic chemicals, but also including compounds that occur naturally that are concentrated in the local environment to toxic levels via human activities. The second is through human-caused changes in naturally occurring chemicals, compounds, or physical parameters (e.g., temperature, turbidity, nutrients, pH). The third is through introduction of new diseases or pathogens, or through other activities that cause an unnatural increase in disease organisms.

Here we treat the these first two of these mechanisms, focusing on the marine and estuarine waters of Puget Sound, and restricting our treatment to degradation caused by human activities. Future editions of the Update will expand the treatment to include pathogens, the condition of fresh water systems, and natural sources of change in water quality.

1. Toxic Contaminants

Background

Determination of the significance of contamination of the Puget Sound ecosystem by toxic chemicals requires measuring the health of organisms, understanding how toxics move through the ecosystem, and estimating the risks posed by exposure to toxic chemicals. In this report we integrate some of the physical characteristics of toxics in the system with the negative effects they could cause on biota. The "threat" of toxics is dealt with separately in Section 3. Here we provide a comprehensive overview of toxics in the system, regardless of their value as an indicator of water quality. Thus, some information presented in this section comes from metrics that may not be the best indicators of water quality, but instead addresses issues of human or ecosystem health (e.g., salmon).

Toxic contaminants have been released into the Puget Sound and its watersheds for decades by human activities. Concern over the possible harmful effects of these pollutants in the ecosystem led to the creation of Washington's Pollution Control Commission in 1945, almost 30 years before the federal Clean Water Act. The Puget Sound Water Quality Authority was established in 1985 to address pressing water quality issues, and by 1989 monitoring and assessment of water quality in Puget Sound had begun with the Puget Sound Ambient Monitoring Program (PSAMP).

The goals of PSAMP included characterizing status and trends of the condition of Puget Sound. Now called the Puget Sound Assessment and Monitoring Program, it currently exists as a consortium of regional scientists from a number of agencies who monitor and assess ecosystem health. Other ongoing toxics monitoring efforts in Puget Sound include MusselWatch (Kimbrough et al. 2008), a national program that has been active in Puget Sound since the 1980s, and King County's Marine Monitoring Program .

The Washington Department of Ecology has evaluated and identified 17 chemicals of concern for Puget Sound (Table 1)(Hart Crowser 2007), based on threat or known harm to biota. Of these, only five chemicals have been banned nation-wide under the Toxic Substances Control Act (TSCA) since 1976. Washington State recently became the first state to ban a class of relatively new chemicals, polybrominated flame retardants (PBDEs), because of human and environmental health concerns.

Table 1. Washington Department of Ecology's list of Chemicals of Concern. Table reprinted from Hart Crowser 2007

Chemical of Concern	Category Addressed	Harm or threat
Arsenic	Arsenic	Associated with sediment toxicity and benthic community impairment
Cadmium	Cadmium	Accumulation in shellfish
Copper	Copper	Associated with sediment toxicity and benthic community impairment; affects salmonids and stream health
Lead	Lead	Associated with sediment toxicity and benthic community impairment
Mercury	Mercury	Target of fish consumption advice; Associated with sediment toxicity and benthic community impairment
Total PCBs (a)	PCBs	Target of fish consumption advice; accumulation in fish, birds, mammals; associated with sediment toxicity and benthic community impairment
Low molecular weight PAHs (b)	PAHs	Liver lesions and reproductive impairment in fish from urban bays; associated with sediment toxicity and benthic community impairment
Carcinogenic PAHs (c)	PAHs	Liver lesions and reproductive impairment in fish from urban bays; associated with sediment toxicity and benthic community impairment
Other high molecular weight PAHs (d)	PAHs	Liver lesions and reproductive impairment in fish from urban bays; associated with sediment toxicity and benthic community impairment
Sum of DDT and metabolites (e)	Pesticides	Accumulation in fish, birds, and mammals; associated with sediment toxicity and benthic community impairment
Triclopyr (f)	Pesticides	Category thought to affect salmonids and stream health
Total dioxin TEQs from dioxins & furans (g)	Dioxins and furans	Accumulation in birds and mammals; furans associated with sediment toxicity and benthic community impairment
bis(2-Ethylhexyl)phthalate	Phthalate esters	Category shown to accumulate in fish, invertebrates, and sediment of urban waterways at levels triggering sediment clean up activities
Total PBDEs (h)	PBDEs	Accumulation in sediments, fish, and harbor seals
Nonylphenol	Hormone disrupting chemicals	Category thought to cause reproductive impairment observed in fish from urban bays
Oil or petroleum product (i)		Kills and reduces fitness of marine organisms
Zinc		Increasing concentrations may threaten aquatic resources

(a) Sum of polychlorinated biphenyl congeners.

(b) Polyaromatic hydrocarbons: acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene (per WAC 173-204-320).

(c) Polyaromatic hydrocarbons: benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, and indeno(1,2,3-c,d)pyrene (per USEPA).

(d) Polyaromatic hydrocarbons: benzo(g,h,i)perylene, fluoranthene, and pyrene (WAC 173-204-320 high molecular weight PAHs not on U.S. EPA list of carcinogenic PAHs).

(e) DDT = Dichlorodiphenyltrichloroethane.

(f) Input from the project team did not reflect consensus to include this compound as currently used pesticide. Other candidates suggested by project team members included diazinon and dichlorbenil.

(g) TEQ = Toxicity equivalent.

(h) PBDEs = Polybrominated diphenyl ethers. Sum of congeners have been normalized.

(i) Specified as crude oil, specific refined product (e.g., diesel, gasoline, heavy fuel oil), or analytical result as TPH-D or TRPH.

Toxic contaminants are considered a priority threat in Puget Sound because they may harm the health of biota. In many cases harm can be difficult to observe; effects can be non-lethal (behavioral) or affect reproductive potential. The status of toxic contaminants in ecosystems typically is reported using a) metrics of exposure, such as the concentration of contaminant residues in tissues; b) health effects such as cancer or reproductive impairment that are known to be caused by such exposure (i.e., are "toxicopathic"); c) concentration of toxics in abiotic media such as sediments or water; d) toxicity of abiotic media; e) benthic infaunal community metrics, or f) an index value calculated from some combination of a-e. The process of "bioconcentration" of toxics from abiotic media to biota is well documented in some cases, suggesting that toxic contaminants in abiotic media can serve as a proxy for or predictor of exposure (Meador 2006).

Measuring toxic contaminants in the environment is expensive and sometimes logistically difficult, so monitoring and assessment studies or programs are challenged with targeting contaminants that pose the greatest threat based on a number of criteria including:

- level of toxicity to organisms
- types of harm caused
- persistence in the environment
- rates of bioaccumulation and biomagnification
- frequency of occurrence in the ecosystem
- spatial distribution in the ecosystem
- threats to specific taxa

Furthermore, the toxicity of a contaminant to an organism depends on the degree to which it is exposed to the chemical. Ideally, status is reported with respect to both the degree of exposure, and the effects (impacts) that exposure causes.

This section summarizes the status and trends of contaminant exposure and effects for key species to four major classes of toxic contaminant. Metrics reported here include: a) measurements of contaminant concentration in organisms' bodies ("tissue residues") or concentration of contaminant metabolites; b) toxicopathic effects (e.g., liver disease and various measures of reproductive impairment); c) concentration of toxics in sediments, primarily as a source of and proxy for biotic exposure; and d) a multimetric toxics-related index of sediment health.

The focus of this section is on toxic contaminants as they relate to biotic exposure and effects. Various species have been used over the years as indicators of toxics status and trends, based on key life history characteristics designed to evaluate the presence, fate, and transport of toxics in the complete food web. Key life history characteristics include:

- Where the organism lives (its habitat, e.g., benthic vs pelagic)
- Trophic level
- Longevity (long lived species have a greater potential for accumulative exposure)
- Migration/residency relative to contaminated habitats
- Prey or food preferences

Furthermore, the focus of this report is limited to the marine ecosystem. Evaluation of loadings and sources, such as from stormwater or atmospheric deposition, is not included.

Toxic contaminants in sediments and fish tissues have been two of the most widely monitored and assessed indicators of ecosystem health in Puget Sound. Understanding the significance of the threat posed by sediment contamination requires an understanding of the relationship between sediment pollution, biotic exposure, and the movement of contaminants from sediments to biota. The majority of data useful for a broad-scale evaluation of status and trends in both sediment and biota comes from the PSAMP long-term monitoring and assessment studies. Results from these efforts have been published primarily in the periodic Puget Sound Update series and in other state agency reports. Most PSAMP data collection methods use vetted protocols (e.g., Puget Sound Estuary Program 1989a (revised), 1989b (Revised), 1990, 1996a, 1996b) which may have been modified over time following internal agency peer review or review among PSAMP principal investigators. Reviews of PSAMP were performed by a panel of external experts in 1995 (Shen 1995) and again in 2005 by PSAMP's Management Committee (PSAMP unpublished). In cases where agency-endorsed or other adequate processes for peer review were performed, and where procedures were vetted as above, PSAMP results from the Puget Sound Update series or other Agency reports are cited or presented here. Data or findings that fail to meet these requirements are omitted.

Status and Trends

Persistent Bioaccumulative Toxics (PBTs)

Persistent bioaccumulative toxic contaminants are a class of substances comprising primarily synthetic chemicals designed and manufactured to meet a wide range of industrial, agricultural, or residential needs. Because they are persistent and bio-accumulative, they are cause for concern when released into the environment. These chemicals generally resist physical, chemical, and metabolic breakdown, so they remain unchanged in the environment for a long period of time. Their concentration increases in the body with chronic or increasing exposure or intake, and they are toxic, causing harm to biota. Because of these characteristics, PBTs have been the focus of intense research world-wide, and large PBT databases exist for risk assessors, modelers, and regulators (Weisbrod et al. 2007).

In Puget Sound marine and estuarine waters the PBTs of primary concern are summarized by Hart Crowser (2007). Those for which broad status information exists in Puget Sound include polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), organo-chlorinated pesticides (OCPs) such as dichlorodiphenyltrichloroethane (DDTs), and mercury. These contaminants have been measured or monitored in a wide range of species in Puget Sound from as early as the mid 1970s to present, with consistent monitoring in several species from 1989 to present. Although polychlorinated dibenzo-p-dioxins (PCDDs), and polychlorinated dibenzofurans (PCDFs) have been detected in English sole from the most heavily contaminated embayment in Puget Sound (Elliott Bay; Sloan and Gries 2008), these compounds are considered a minor threat to apex predators such as harbor seals in Puget Sound (Ross et al. 2004) that could otherwise potentially be exposed to high PBT levels via bio-magnification.

Perhaps the clearest PBT exposure-effects relationship of concern in the Puget Sound marine waters is the exposure of apex predators such as Southern Resident Killer Whales (SRKW) and harbor seals to PCBs and PBDEs (Figure 1). Hickie et al. (2007) and Ross et al. (2004) reported PCB exposure in harbor seals from Puget Sound at levels predicted to impair health, while Ross et al. (2000) described the SRKW population as the most PCB-contaminated of all cetaceans in the world. Calculations made by Hickie et al. (2007) and Ross et al. (2004) suggested that during their years of peak exposure, all members of the SRKW population were affected, and that exposure exceeded thresholds by 3 to 31 times. The authors estimated that based on PCBs alone, it would take until the year 2089 for 95% of the population to drop below the health effects threshold, given current PCB trends. Such PBT contamination is considered a risk to recovery of this population (Krahn et al. 2002).

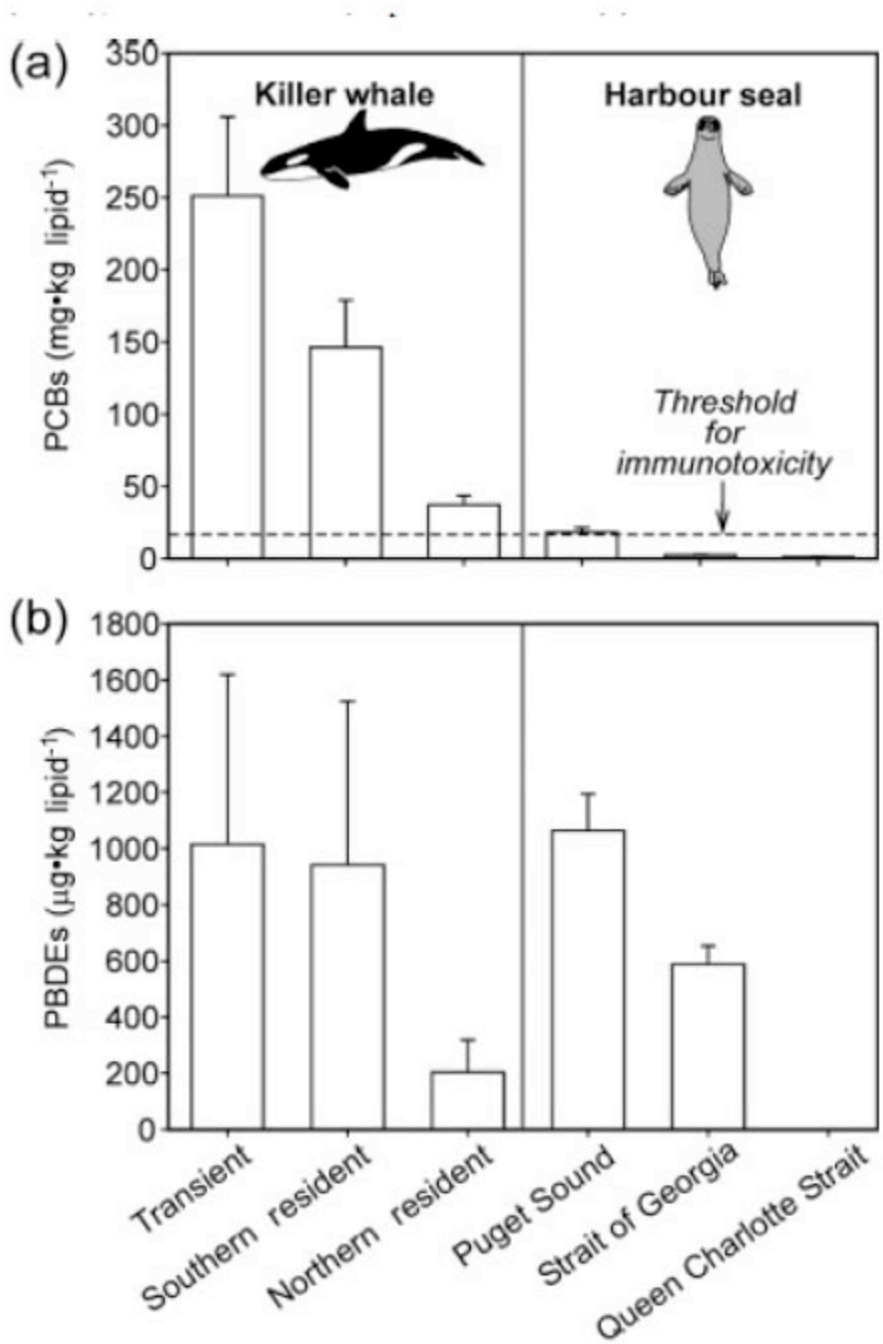


Figure 1. Persistent bioaccumulative toxics (PCBs and PBDEs) in two apex predators from the Puget Sound and Strait of Georgia, with health effects threshold for PCBs. Reprinted with permission from Ross (2006)

The source of PCBs to these animals is their food, primarily chinook salmon for killer whales (e.g., Krahn et al. 2007) and a mix of small pelagic and benthic fish for harbor seals (Cullon et al. 2005). O'Neill and West (2009) reported high PCB body burdens in chinook salmon that reside in Puget Sound, compared to more oceanic migrants (Figure 2) and West et al. (2008) reported high PBC burdens in Pacific herring from Central and Southern Puget Sound, compared with Southern Strait of Georgia and with herring from highly polluted regions of the Baltic Sea (Figure 3). This illustrates the importance of PBT transfer via trophic interactions and the need to understand PBT fate and transport processes in food webs.

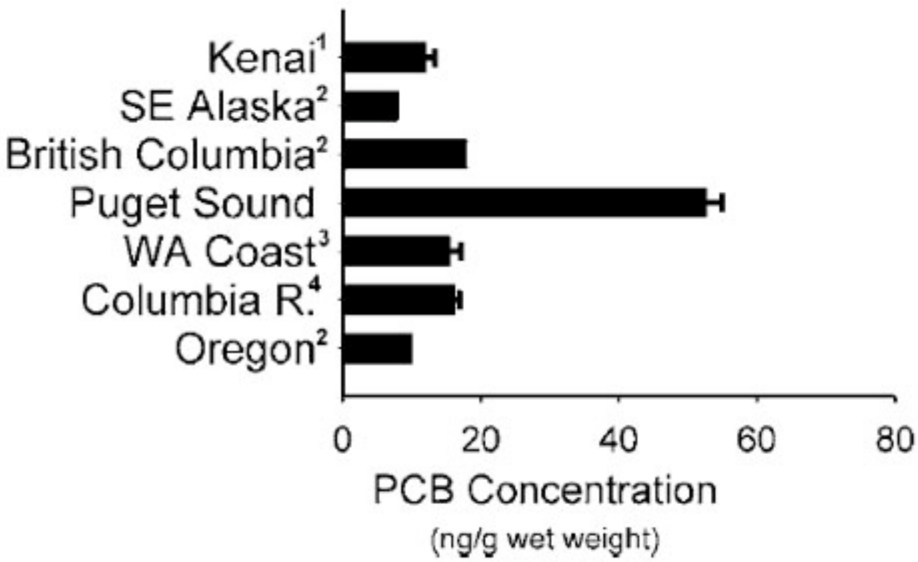


Figure 2. Comparison of PCB tissue residues in adultChinook salmon returning to spawn in Puget Sound and Pacific Oceanic coastal rivers. See West and O’Neill 2009 for a description of sampling location and full data citations. All samples were from adult Chinook salmon returning to natal rivers to spawn. Copyright American Fisheries Society. Used with permission.

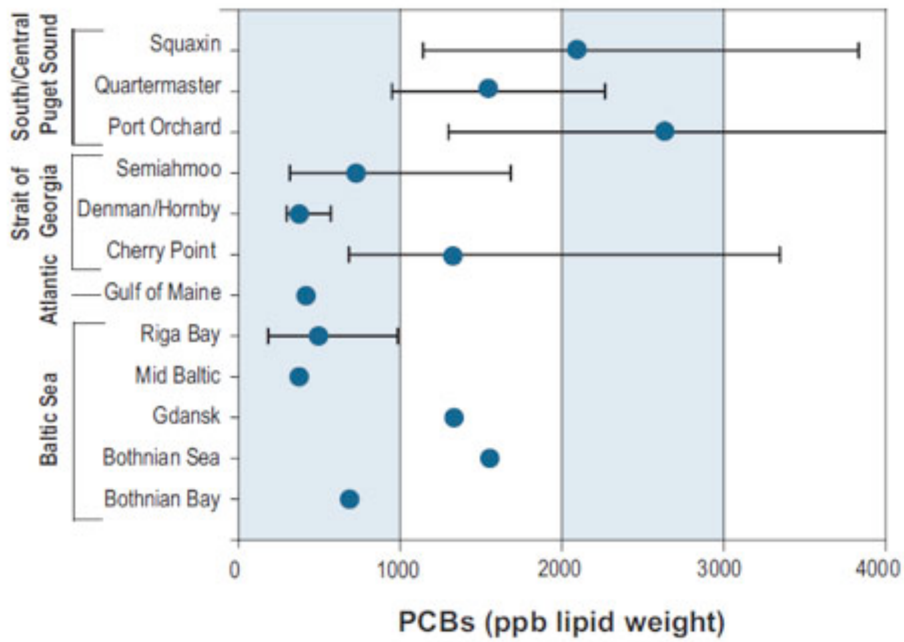


Figure 3. Comparison of PCBs among six populations of Pacific herring Puget Sound and the Georgia Basin, and Atlantic and Baltic herring. Squaxin population from South Puget Sound, Quartermaster Harbor and Port Orchard from Cental Puget Sound, and Semiahmoo, Denman/Hornby, and Cherry Point from the Southern Strait of Georgia (Reprinted from 2007 Puget Sound Update; data from West et al. 2008)

PCB exposures in chinook salmon pose a health risk to the fish, as well as to the humans that consume them. PCBs in chinook salmon(O'Neill and West 2009) exceeded an effects threshold reported by Meador (2002), indicating a threat to normal growth and maturation processes for these salmon. Furthermore, the Washington Department of Health has issued guidelines that recommend restrictions to dietary intake of these fish to protect human health .

In sediments, PBTs tend to accumulate in industrial or urbanized habitats near their sources, prompting focused attention on toxics there (Partridge et al. 2009, Puget Sound Estuary Program 1988). The Environmental Protection Agency's Superfund program has focused sediment cleanup efforts in a number of Puget Sound's urbanized embayments since 1980 (2007 Puget Sound Update). Overall, however, Ecology's long-term PSAMP efforts (methods peer reviewed: Dutch et al. 2009) have reported PCB levels in sediments exceeding Washington State Sediment Quality Standards (SQS) in only 19 of over 500 stations from the full extent of Puget Sound sampled between 1997 and 2008. Washington State Sediment Quality Standards, adopted as part of Washington's environmental regulations, define levels at which various chemicals present in sediments become harmful to marine life (WAC 173-204). All PCB exceedances were located in sediments taken from urban embayments in the Central Puget Sound basin. Data indicate that PCB concentrations in Elliott Bay sediments, where most of the exceedances have occurred, have been declining (Partridge et al. 2009). A Washington State Sediment Quality Standard does not yet exist for PBDEs, and although PBDE concentrations were lower than PCBs overall, they were concentrated in Central Puget Sound and its urbanized embayments.

Long-term Sound-wide monitoring efforts have shown that this urban PCB and PBDE sediment signal is reflected in benthic (bottom-dwelling) and demersal (near-bottom) species. Tissue residues of PCBs and PBDEs were greatest in English sole (benthic), rockfish (demersal) and lingcod (demersal) from Elliott Bay, Commencement Bay, and Sinclair Inlet, or from other Central Puget Sound urban or near-urban locations (as reported in 2007 Puget Sound Update). PCB residues in blue mussels were greatest in Central Puget Sound locations (Kimbrough et al. 2008). These studies demonstrate the relationship between benthic (or benthic-feeding) species and the contaminant-condition of their environment.

Although pelagic (open-water) species may not have direct trophic connections with the sediment-contaminated benthic food web, pelagic food web species in urbanized waters exhibited high levels of exposure to PBTs. Pacific herring (West et al. 2008), Chapter2a.Salmonids#chinookanchor|chinook salmon]] (O'Neill and West 2009), and harbor seals (Ross et al. 2004) that reside in Puget Sound conform to this pattern. PCB and PBDE tissue residues were consistently greatest in individuals of these three species from the Central or Southern Puget Sound Basins, compared with conspecifics from the Strait of Georgia, Strait of Juan de Fuca, or Pacific Ocean. As noted previously, PCB and PBDE tissue residues exceeded health effects thresholds in salmon] and [[Chapter2a.HarborSeals|harbor seals.

PBDEs have only relatively recently been added to tissue monitoring and assessment programs in Puget Sound. Using archived tissue samples, West and O'Neill (2007) observed 80 ng/g Total PBDEs (wet wt) in herring from Central Puget Sound in 2001, roughly one-half the concentration of Total PCBs reported for the same samples from (West et al. 2008).

Polycyclic Aromatic Hydrocarbons (PAHs)

PAHs are derived from fossil fuels, and are typically produced via combustion of these fuels (pyrogenic) or occur as constituents of petroleum (petrogenic). Most of these chemicals exist naturally, but their presence in the environment becomes problematic when they are concentrated to toxic levels by human activities. Many PAHs are persistent in the environment, however they are typically metabolized by vertebrates when exposed to relatively low concentrations, and therefore do not tend to accumulate in their bodies. For this reason, food web magnification of PAHs for apex predators is of less concern than for PBTs.

However, both exposure and effects measures from biota indicate that PAHs represent a serious threat to the health of some Puget Sound biota. PAHs in blue mussels from seven of 14 sites in Puget Sound waters were termed “high” (at or above the 85th percentile for all 263 stations nationwide in at least half the years sampled between 1986 and 1991) by the national Mussel Watch Program (O'Connor 2002). Currently, the PAH status of mussels from eight of 10 stations in Puget Sound is rated either “medium” or “high” (Kimbrough et al. 2008), with a number of locations that met or exceeded comparable mussel samples taken in highly urbanized areas of the Baltic Sea. Tissue residues of PAHs in blue mussels could originate from capturing and consuming PAH-laden particles derived from nearby sediments (Baumard et al. 1999). This hypothesis is supported by the observation that PAHs in Puget Sound mussels are typically greatest in urban sites (Kimbrough et al. 2008).

Because PAHs are metabolized by vertebrates, measuring the exposure of fish, birds and mammals is less straightforward than measuring PBT tissue residues. Metabolites of PAH compounds can be measured in fish bile (Krahn et al. 1984), and these so-called biliary Fluorescing Aromatic Compounds (FACs) have been monitored in English sole, rockfish and herring as a semi-quantitative measure of PAH exposure in these species in Puget Sound (West et al. 2001). In the benthic or demersal fish species biliary FACs were consistently greatest in fish taken from urbanized embayments.

PAHs are linked to a number of toxicopathic fish diseases. English sole develop degenerative liver disease when exposed to PAHs in the sediments where they feed (Myers et al. 1990, Myers et al. 1991). Other effects include interruption in growth, and reproductive impairments (Johnson et al. 2002). Myers et al. (2008) documented the recovery of health among English sole in Eagle Harbor, a highly PAH-contaminated Superfund site, where prevalence of PAH-induced liver disease dropped from 80% to 5% over a ten year period during remediation, which included sequestration of PAHs with sediment capping (Figure 4).

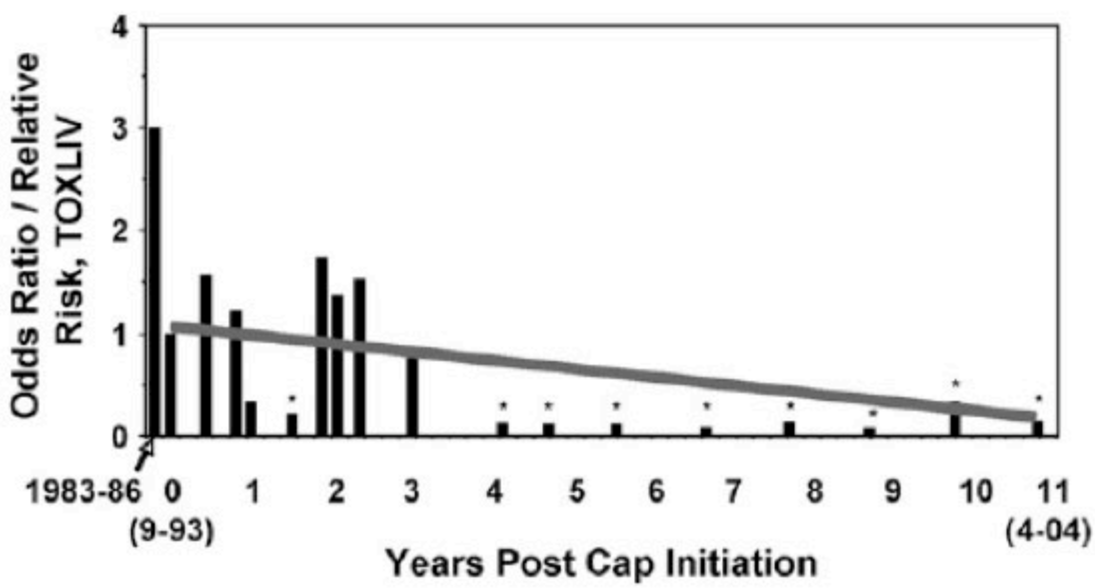


Figure 4. Temporal trend in liver disease of English sole from a sediment-remediated site in Eagle Harbor, Washington. Reprinted with permission from Myers et al. 2008.

PAH-induced liver disease has been tracked in English sole by PSAMP for 20 years in eight Puget Sound locations. This tracking study uses protocols developed to monitor histopathological health metrics in fish, including toxicopathic liver disease (Puget Sound Estuary Program 1987), which are regularly reviewed by a contract pathologist. Recent results from this tracking study, reported in the 2007 Puget Sound Science Update, include the following:

- liver disease in English sole is associated primarily with Puget Sound's highly contaminated urban embayments near Seattle (Elliott Bay), Tacoma (Thea Foss Waterway), and Everett (Port Gardner). The risk of developing liver disease in these areas was two to six times that of normal background
- the risk of developing PAH-induced liver disease has remained low and unchanged at six of eight long-term stations, and has declined significantly in English sole from Elliott Bay (Seattle).
- high molecular weight PAHs, the group most often associated with liver disease (Myers et al. 1991), have declined in Elliott Bay sediments (Partridge et al. 2009), and in English sole bile from Elliott Bay

Pelagic fish in Puget Sound have also exhibited exposure to PAHs. Pacific herring, a small, schooling pelagic planktivorous fish, have consistently exhibited PAH exposures in Central Puget Sound similar in magnitude to benthic (English sole) and demersal (rockfish) species from most urbanized embayments for the past 10 years (2007 Puget Sound Science Update).

The source of persistent organic pollutants such as PAHs in adult fishes is widely thought to be dietary and because PAHs are metabolized, their biliary FAC measurements tend to reflect PAH loads in prey that have been consumed recently. Pacific herring is a fully pelagic species that consumes primarily zooplankton prey, with little obvious trophic connection to contaminated sediments in Puget Sound. Small schooling pelagic planktivores such as herring may accumulate PAHs that have originally been ad- or absorbed to plankton, and then magnified among planktonic invertebrates in the food web (Wolfe et al. 2001). It has been suggested that some PAHs loaded from atmospheric deposition or other sources enter the pelagic food web directly via bioaccumulation by phytoplankton, and then are magnified through the planktonic food chain to their fish predators (Larsson et al. 2000). In Puget Sound, this may explain why pelagic species far removed, both trophically and spatially, from contaminated sediments exhibit such high exposure to PAHs, and could help to inform decisions regarding how to mitigate exposure of biota to PAHs in Puget Sound.

PAH exposure may pose a significant threat to sensitive life stages of Puget Sound biota. Links between chronic, sublethal PAH levels and health impacts in fish embryo and larval stages, as well as delayed population declines from early-life PAH exposures have been well established (Carls and Meador 2009, Peterson et al. 2003). In addition, PAHs from creosote, such as found on treated pilings, can kill embryos (Vines et al. 2000). Herring embryos exhibiting chronic mortality from at least one spawning ground in Puget Sound have experienced exposure to PAHs exceeding a PAH effects threshold (as reported in 2007 Puget Sound Science Update), however a PAH cause-and-effect link to mortality has not yet been established in Puget Sound.

Contaminants of Emerging Concern (CECs)

This group of contaminants comprises a broad range of chemical classes whose adverse effects on biota is only recently becoming apparent. They range widely in solubility, persistence, toxicity, and mode-of-action, and include such classes as perfluorinated compounds (from the creation of fluoropolymers, semiconductors, and fire-fighting foam), nonylphenol (a surfactant), bisphenol-A and phthalates, both used in plastics, and pharmaceuticals and personal care products. Many of these contaminants have endocrine disrupting capacity, and so may be reported as Endocrine Disrupting Compounds (EDCs); some are specifically estrogenic and so may be reported as xenoestrogens. Although some of these contaminants have been measured in Puget Sound fishes (West et al. 2001), and some are monitored in Puget Sound sediments (Dutch et al. 2009), many CECs currently are not measured in environmental media on a broad scale (Muir and Howard 2006). Moreover, analytical techniques for measuring tissue residues or metabolites for many CECs are lacking. Two CECs, nonylphenol (NNP) and bis(2-ethylhexyl)phthalate (DEHP) are included in Washington Department of Ecology's Chemicals of Concern list (Hart Crowser 2007).

In Puget Sound sediments, at least one phthalate-chemical, DEHP, exceeded the Washington State sediment quality standard, and appears to be increasing (Partridge et al. 2009). DEHP is associated with a wide range of toxicopathic disease including endocrine disruption (e.g., feminization of males). English sole in Puget Sound have shown evidence of exposure to xenoestrogenic compounds, even though the causative pollutants remain unknown. Johnson et al. (2008) reported the presence of vitellogenin, a precursor to egg protein normally found only in females, in male English sole from twelve of sixteen locations sampled in Puget Sound. Moreover, both females and males from one affected population in Elliott Bay exhibited altered reproductive timing, possibly related to the unknown estrogenic pollutants (Figure 5). Based on spatial patterns in the fish impairment, these authors hypothesized that the source of xenoestrogens to these fish was industrial discharges, surface runoff, or sewage, and discussed the most likely causative pollutants.

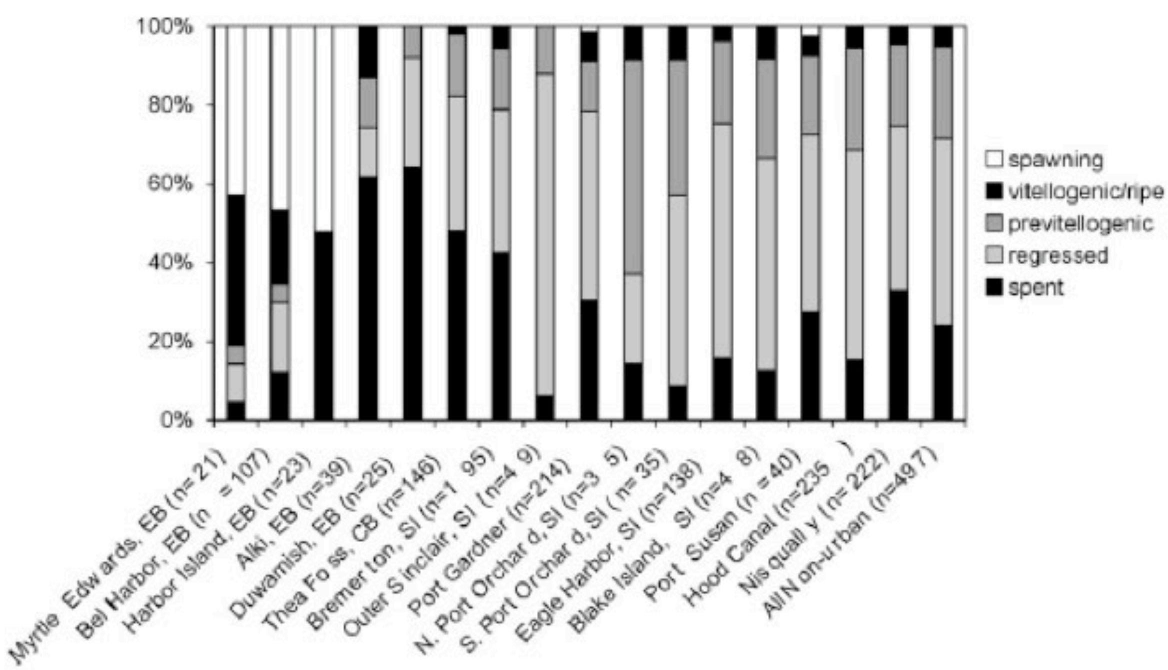


Figure 5. Unusual reproductive timing in female English sole from three Elliott Bay sites compared to 14 other Puget Sound locations. Reprinted with permission from Johnson et al. 2008.

Metals

Like PAHs, metals occur naturally in the environment. Metals become contaminants of concern when they are altered chemically or redistributed in the environment in ways that make them more available or toxic to biota. In some cases (e.g., mercury) metals may naturally occur in biota in a magnitude great enough to cause concern for humans consuming them (Barghigiani and DeRanieri 1992).

Metals have been monitored Sound-wide in sediments (Dutch et al. 2009) and fish tissue (West et al. 2001) since 1989, and in blue mussels since 1986 (Kimbrough et al. 2008). Metal accumulation in mussels has been unremarkable, except that the greatest tissue residues of mercury, nickel, and lead occurred in highly urbanized areas, suggesting anthropogenic contributions. "Medium" concentration of a number of metals was reported from locations with the greatest exposure to oceanic waters, far removed from human activities, suggesting accumulation of natural sources.

The Sediment Quality Triad

The Sediment Quality Triad Index (SQTI) is a multi-metric index developed to describe the degradation of sediment condition by toxic contaminants (Chapman 1990). Because the SQTI incorporates toxic contaminants from a broad range of classes, it is presented separately in this section. The SQTI combines the results from pollutant concentration measures, toxicity studies (exposing sensitive organisms to sediments or their extracts), and analysis of the composition of the infaunal community in sediments. This last measure is typically based on best professional judgment, and integrates nine different measures of community structure and the presence/absence of pollutant tolerant/sensitive species. A substantial advantage of the SQTI is that it examines both effects and exposure metrics, and uses a weight-of-evidence approach to integrate three important measures of sediment quality into one indicator that can be compared Puget Sound-wide.

SQTI has been used extensively in Puget Sound as an indicator of sediment health (Long et al. 2003). A seven-year comprehensive SQTI survey of 381 Puget Sound sediment stations reported that although only one percent of sediments were "degraded", most of these sediments were in highly productive shallow-water embayments or river deltas (reported in 2007 Puget Sound Science Update). Thirty-eight percent of sediments in Puget Sound were considered of "intermediate" quality, wherein degradation was detected in one or two of the three SQTI metrics. A full, final report for this study is currently being reviewed by the Washington Department of Ecology.

Uncertainties.

One important uncertainty concerns the linkage between the status and trends of toxic contaminants in the ecosystem and the associated population-level effects on biota. Health-effects thresholds are lacking for the great majority of toxic contaminants monitored in Puget Sound, especially for complex mixtures of chemicals. Constructing models that predict population-level effects from lethal or sub-lethal effects of single contaminants or mixtures is a challenge, because such models often carry a great deal of uncertainty that can result in wide-ranging outcomes. Except for oil spills or other episodic events, observations of mortality directly attributable to toxic contaminants are rare. A singular exception to this in Puget Sound is pre-spawning mortality of coho salmon in urban streams (Collier et al. 2004), a phenomenon widely attributable to road-based contaminants. In this case Spromberg and Scholz (2009) predict extirpation of coho spawning runs over decadal time scales in urbanized streams.

In addition, recent findings on the susceptibility of eggs and larvae of Pacific herring to fossil fuel-derived PAH compounds (Carls et al. 1999, Incardona et al. 2009) combined with field studies that demonstrate exposure of their embryos to PAHs in Puget Sound (as reported in 2007 Puget Sound Update) show the risk of mortality to this sensitive life stage from exposure to chemical pollutants in Puget Sound. However, because such mortalities are extremely difficult to observe or measure in the wild, they currently are not used in decision-making.

Uncertainties unique to status and trends of monitoring data include shifting methodologies and study designs over long time periods. For example, PCBs reported from some studies in this document have been analyzed using a range of methodologies including mixtures (Aroclor) analysis and congener-based methods. Careful evaluation of all methods, including those for biological covariates, must be made when comparing these data across studies, and when applying threshold criteria.

Summary

Human activities have resulted in the introduction or elevation of toxic contaminants into Puget Sound. These include Persistent Bio-accumulative Toxics such as PCBs, PBDEs and DDTs, chemicals derived from fossil-fuels (PAHs), various metals, and Contaminants of Emerging Concern, including Endocrine Disrupting Compounds and pharmaceuticals and personal care products. In Puget Sound, a number of PBT chemicals are present in apex predators such killer as whales and harbor seals and in their primary food sources (salmon and herring) in concentrations that may harm their health and impair recovery of populations that are depressed. For most PBTs, the highest concentrations occur in sediments and biota from the Central Puget Sound and its urbanized embayments, or localized urbanized shorelines in other Puget Sound basins. PAH monitoring of mussel tissue has caused Puget Sound to be characterized as a hot spot for that class of contaminants, relative to other urban areas in the nation. PAH chemicals have also been detected in fish bile and identified as a causative factor in liver disease in English sole in Puget Sound waters. Juvenile life stages of fish may be particularly susceptible to PAH toxicity. Reproductive effects of endocrine-disrupting compounds have been detected in benthic Puget Sound fish but the consequences of exposure at the population level and long-term trends are not known.

Literature Cited

Barghigiani, C., and DeRanieri, S. 1992. Mercury content in different size classes of important edible species of the Northern Tyrrhenian Sea. Marine Pollution Bulletin 24(2): 114-116.

Baumard, P., Budzinski, H., Garrigues, P., Narbonne, J.F., Burgeot, T., Michel, X., and Bellocq, J. 1999. Polycyclic aromatic hydrocarbon (PAH) burden of mussels (*Mytilus* sp.) in different marine environments in relation with sediment PAH contamination, and bioavailability. Marine Environmental Research 47(5): 415-439.

Carls, M.G., and Meador, J.P. 2009. A Perspective on the toxicity of petrogenic PAHs to developing fish embryos related to environmental chemistry. Human and Ecological Risk Assessment: An International Journal 15(6): 1084 - 1098.

Carls, M.G., Rice, S.D., and Hose, J.E. 1999. Sensitivity of fish embryos to weathered crude oil: Part I. Low-level exposure during incubation causes malformations, genetic damage, and mortality in larval Pacific herring (*Clupea pallasi*). Environmental Toxicology and Chemistry 18(3): 481-493.

Chapman, P.M. 1990. The sediment quality triad approach to determining pollution-induced degradation. Science of The Total Environment 97-98: 815-825.

Collier, T.K., Labenia, J., Myers, M.S., and Scholz, N.L. 2004. Abstracts from the twelfth international symposium on pollutant respsonses in marine organisms - mechanisms of toxicity - organics: Extreme PRIMO: premature mortality of adult coho salmon associated with stormwater discharges. Marine Environmental Research 58(2-5): 359-369.

Cullon, D.L., Jeffries, S.J., and Ross, P.S. 2005. Persistent organic pollutants in the diet of harbor seals (Phoca vitulina) inhabiting Puget Sound, Washington (USA), and the Strait of Georgia, British Columbia (Canada): a food basket approach. Environmental Toxicology and Chemistry 24(10): 2562-2572.

Dutch, M., Partridge, V., Weakland, S., Welch, K., and Long, E. 2009. Quality Assurance

Project Plan: The Puget Sound Assessment and Monitoring Program Sediment Monitoring Component. Washington State Department of Ecology. Olympia, WA

Hart Crowser. 2007. Control of Toxic Chemicals in Puget Sound. Phase 1: Initial estimate of loadings. Washington Department of Ecology. Olympia, WA

Hickie, B.E., Ross, P.E., MacDonald, R.W., and Ford, J.K.B. 2007. Killer whales (*Orcinus orca*) face protracted health risks associated with lifetime exposure to PCBs. Environmental Science and Technology 41(18): 6613-6619.

Incardona, J.P., Carls, M.G., Day, H.L., Sloan, C.A., Bolton, J.L., Collier, T.K., and Scholz, N.L. 2009. Cardiac Arrhythmia Is the Primary Response of Embryonic Pacific Herring (Clupea pallasi) Exposed to Crude Oil during Weathering. Environmental Science & Technology 43(1): 201-207.

Johnson, L., Collier, T.K., and Stein, J.E. 2002. An analysis in support of sediment quality thresholds for polycyclic aromatic hydrocarbons (PAHs) to protect estuarine fish. Aquatic Conserv: Mar. Freshw. Ecosyst. 12: 517-538.

Johnson, L.L., Lomax, D.P., Myers, M.S., Olson, O.P., Sol, S.Y., O'Neill, S.M., West, J., and Collier, T.K. 2008. Xenoestrogen exposure and effects in English sole (*Parophrys vetulus*) from Puget Sound, WA. Aquatic Toxicology 88(1): 29-38.

Kimbrough, K.L., Lauenstein, G.G., Christensen, J.D., and Apeti, D.A. 2008. An assessment of two decades of contaminant monitoring in the Nation's Coastal Zone. National Oceanic and Atmospheric Administration/National Center for Ocean Science Technical Memorandum, Silver Spring, MD.

Krahn, M.M., Hanson, M.B., Baird, R.W., Boyer, R.H., Burrows, D.G., Emmons, C.K., Ford, J.K.B., Jones, L.L., Noren, D.P., Ross, P.S., Schorr, G.S., and Collier, T.K. 2007. Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from Southern Resident killer whales. Marine Pollution Bulletin 54(12): 1903-1911.

Krahn, M.M., Wade, P.R., Kalinowski, S.T., Dalheim, M.E., Taylor, B.L., Hanson, M.B., Ylitalo, G.M., Angliss, R.P., Stein, J.E., and Waples, R.S. 2002. Status Review of Southern Resident Killer Whales (*Orcinus orca*) under the Endangered Species Act. Edited by U.S.D.o. Commerce. p. 133.

Krahn, M.M., Myers, M.S., Burrows, D.G., and Malins, D.C. 1984. Determination of metabolites of xenobiotics in the bile of fish from polluted waterways. Xenobiotica 14(8): 633-646.

Larsson, P., Andersson, A., Broman, D., Nordbäck, J., and Lundberg, E. 2000. Persistent organic pollutants (POPs) in pelagic systems. Ambio 29(4-5): 202-209.

Long, E.R., Dutch, M., Aasen, S., Welch, K., and Hameedi, M.J. 2005. Spatial extent of degraded sediment quality in Puget Sound (Washington State, U.S.A.) based upon measures of the sediment quality triad. Environmental Monitoring and Assessment 111: 173-222.

Long, E.R., Dutch, M., Aasen, S., Welch, K., and Hameedi, M.J. 2003. Chemical contamination, acute toxicity in laboratory tests, and benthic impacts in sediments of Puget Sound: a summary of results of the joint 1997-1999 Ecology/NOAA survey. Publication Number 03-03-049, or Technical Memorandum NOS NCCOS CCMA 163, Washington Department of Ecology, Lacey, WA.

Meador, J. 2006. Rationale and Procedures for Using the Tissue-Residue Approach for Toxicity Assessment and Determination of Tissue, Water, and Sediment Quality Guidelines for Aquatic

Organisms. Human and Ecological Risk Assessment: An International Journal 12(6): 1018 - 1073.

Meador, J.P. 2002. Use of tissue and sediment-based threshold concentrations of polychlorinated biphenyls (PCBs) to protect juvenile salmonids listed under the US Endangered Species Act. Aquatic Conservation: Marine and Freshwater Ecosystems 12: 493-516.

Muir, D.C.G., and Howard, P.H. 2006. Are There Other Persistent Organic Pollutants? A Challenge for Environmental Chemists. Environmental Science & Technology 40(23): 7157-7166.

Myers, M.S., Anulacion, B.F., French, B.L., Reichert, W.L., Laetz, C.A., Buzitis, J., Olson, O.P., Sol, S., and Collier, T.K. 2008. Improved flatfish health following remediation of a PAH-contaminated site in Eagle Harbor, Washington. Aquatic Toxicology 88: 277-288.

Myers, M.S., Landahl, J.T., Krahn, M.M., Johnson, L.L., and McCain, B.B. 1990. Overview of studies on liver carcinogenesis in English sole from Puget Sound; Evidence for a xenobiotic chemical etiology I: Pathology and epizootiology. Science of the Total Environment 94: 33-50.

Myers, M.S., Landahl, J.T., Krahn, M.M., and McCain, B.B. 1991. Relationships between hepatic neoplasms and related lesions and exposure to toxic chemicals in marine fish from the U.S. West Coastal. Environmental Health Perspectives 90: 7-15.

O'Connor, T.P. 2002. National distribution of chemical concentrations in mussels and oysters in the USA. Marine Environmental Research 53(2): 117-143.

O'Neill, S.M., and West, J.E. 2009. Marine Distribution, Life History Traits, and the Accumulation of Polychlorinated Biphenyls in Chinook Salmon from Puget Sound, Washington. Transactions of the American Fisheries Society 138(3): 616-632.

Partridge, V., Weakland, S., Long, E., Welch, K., Dutch, M., and Jones, M. 2009. Urban Waters Initiative, 2007; Sediment Quality in Elliott Bay. Washington Department of Ecology. Olympia, WA

Peterson, C.H., Rice, S.D., Short, J.W., Esler, D., Bodkin, J.L., Ballachey, B.E., and Irons, D.B. 2003. Long-Term Ecosystem Response to the Exxon Valdez Oil Spill. Science 302(5653): 2082-2086.

Puget Sound Action Team. 2007. Puget Sound Update: Ninth Report of the Puget Sound Assessment and Monitoring Program. Seattle, WA

Puget Sound Estuary Program. 1987. Recommended protocols for fish patholgy studies in Puget Sound. In Recommended Protocols and Guidelines for Measuring Selected Environmental Variables in Puget Sound. Prepaired by Tetra Tech, Inc., Bellevue, WA for U.S. Environmental Protection Agency, Region 10, Seattle, WA. (Looseleaf).

Puget Sound Estuary Program. 1988. Elliott Bay Action Program: Analysis of Toxic Problem Areas prepared for US Environmental Protection Agency, Region X, Seattle, WA.

Puget Sound Estuary Program. 1989a (revised). Recommended guidelines for measuring organic compounds in Puget Sound sediment and tissue samples. In Recommended Protocols and Guidlines for Measuring Selected Environmental Variables in Puget Sound. Prepared by PTI Environmental Services, Bellevue, WA. for U.S. Environmental Protection Agency, Region 10, Seattle, WA (Looseleaf).

Puget Sound Estuary Program. 1989b (Revised). Recommeded guidelines for measuring inorganic compounds in Puget Sound sediment and tissue samples. In Recommended Protocols and Guidelines for Measuring Selected Environmental Variables in Puget Sound. Prepaired by PTI Environmental Services, Bellevue, WA. (Looseleaf) for U.S. Environmental Protection Agency, Region 10, Seattle, WA.

Puget Sound Estuary Program. 1990. Recommended guidelines for sampling demersal fish. In Recommended Protocols and Guidelines for Measuring Selected Environmental Variables in Puget Sound. Prepaired by PTI Environmental Services, Bellevue, WA. for

U.S. Environmental Protection Agency, Region 10, Seattle, WA (Looseleaf).

Puget Sound Estuary Program. 1996a. Recommended guidelines for measuring metals in Puget Sound marine water, sediment and tissue samples, Prepared for the U.S. Environmental Protection Agency, Region 10, Seattle, WA, and Puget Sound Water Quality Authority by King County Environmental Lab, Seattle, WA, Seattle, WA. (Looseleaf).

Puget Sound Estuary Program. 1996b. Recommended guidelines for measuring organic compounds in Puget Sound marine water, sediment and tissue samples, Prepared for the U.S. Environmental Protection Agency, Region 10, Seattle, WA, and Puget Sound Water Quality Authority by King County Environmental Lab, Seattle, WA, Seattle, WA. (Looseleaf).

Ross, P.S., Ellis, G.M., Ikonomou, M.G., Barrett-Lennard, L.G., and Addison, R.F. 2000. High PCB concentrations in free-ranging Pacific killer whales, *Orcinus orca*: effects of age, sex and dietary preference. Marine Pollution Bulletin 40: 504-515.

Ross, P.S., Jeffries, S.J., M.B.Yunker, Addison, R.F., Ikonomou, M.G., and Calambokidis, J.C. 2004. Harbor seals (Phoca vitulina) in British Columbia, Canada, and Washington State, USA reveal a combination of local and global polychlorinated biphenyl, dioxin, and furan signals. Environmental Toxicology and Chemistry 23(1): 157-165.

Ross, P.S. 2006. Fireproof killer whales (*Orcinus orca*): flame retardant chemicals and the conservation imperative in the charismatic icon of British Columbia, Canada. Canadian Journal of Fisheries and Aquatic Science 63: 224-234.

Shen, G.T. 1995. Panel Findings and Recommendations: Based on the Final Comprehensive Review of the Puget Sound Ambient Monitoring Program, Puget Sound Water Quality Authority, P.O. Box 40900, Olympia, WA 98504-0900, Olympia, WA.

Sloan, J., and Gries, T. 2008. Dioxins, furans, and other contaminants in surface sediment and English sole collected from greater Elliott Bay (Seattle). Washington Department of Ecology. Olympia, WA

Spromberg, J.A., John, B.M., and Landis, W.G. 1998. Metapopulation dynamics: Indirect effects and multiple distinct outcomes in ecological risk assessment. Environmental Toxicology and Chemistry 17(8): 1640-1649.

Vines, C.A., Robbins, T., Griffin, F.J., and Cherr, G.N. 2000. The effects of diffusible creosote-derived compounds on development in Pacific herring (*Clupea pallasi*). Aquatic Toxicology 51(2): 225-239.

Weisbrod, A.V., Burkhard, L.P., Arnot, J., Mekenyan, O., Howard, P.H., Russom, C., Boethling, R., Sakuratani, Y., Traas, T., Bridges, T., Lutz, C., Bonnell, M., Woodburn, K., and Parkerton, T. 2007. Workgroup Report: Review of Fish Bioaccumulation Databases Used to Identify Persistent, Bioaccumulative, Toxic Substances. Environmental Health Perspectives 115(2): 255-261.

West, J.E., O'Neill, S.M., Lippert, G.R., and Quinnell, S.R. 2001. Toxic contaminants in marine and anadromous fish from Puget Sound, Washington: Results from the Puget Sound Ambient Monitoring Program Fish Component, 1989-1999. Technical Report. FTP01-14, Washington Department of Fish and Wildlife, Olympia, WA.

West, J.E., O'Neill, S.M., and Ylitalo, G.M. 2008. Spatial extent, magnitude, and patterns of persistent organochlorine pollutants in Pacific herring (*Clupea pallasi*) populations in the Puget Sound (USA) and Strait of Georgia (Canada). Science of the Total Environment 394(2-3): 369-378.

Wolfe, M.F., Schwartz, G.J.B., Singaram, S., Mielbrecht, E.E., Tjeerdema, R.S., and Sowby, M.L. 2001. Influence of dispersants on the bioavailability and trophic transfer of petroleum hydrocarbons to larval topsmelt (*Atherinops affinis*). Aquatic Toxicology 52(1): 49-60.

Harmful Algal Blooms

Background

Rapid growth and accumulation of phytoplankton or other algae can cause algal blooms. Bloom-forming algae that have harmful effects on people or wildlife are commonly termed harmful algal blooms (HABs). In Puget Sound, HABs may be caused by phytoplankton such as dinoflagellates of the genus *Alexandrium*, diatoms of the genus *Psuedo-nitzchia*, raphidophytes of the genus *Heterosigma* or by ulvoid seaweeds. Suspension-feeding bivalves, such as mussels, clams and oysters, can accumulate biotoxins to dangerous levels during HAB events, leading to illness such as paralytic shellfish poisoning (PSP) or amnesic shellfish poisoning (ASP) when the shellfish are ingested by humans, marine mammals and marine birds (Nishitani and Chew 1984, Hallegraeff 1993).

The Washington Department of Health(WDOH) Office of Shellfish Safety and Water Protection regularly monitors biotoxin levels in both recreational and commercial shellfish areas in Puget Sound. The Washington State Public Health Laboratory supports the WDOH through the analysis of shellfish samples. When high levels are detected in sample tissues, shellfish harvest areas are closed in order to protect shellfish consumers from biotoxin-related illness. Closures can have significant effects on commercial, recreational, and subsistence harvest. Episodes of high biotoxin levels are currently unpredictable in time or space due to the interaction of multiple poorly understood environmental factors (Moore et al. 2009).

Paralytic Shellfish Poisoning

Seasonal restrictions on commercial and recreational shellfish harvest due to PSP, sometimes known as "red tide", are common in Washington. The biotoxin that causes PSP temporarily interferes with the transmission of nerve impulses in warm-blooded animals. Symptoms of PSP in humans range from nausea, vomiting, numbness of the lips and tongue and muscle paralysis to death by cardio-respiratory arrest. There is no known antidote for the toxin, and cooking does not destroy the toxin.

Several microscopic organisms that naturally exist in marine water produce the PSP toxin. The species that causes PSP in Washington marine waters is the dinoflagellate *Alexandrium catenella* (Determan et al. 2001). Alexandrium is typically present in small numbers; however, when environmental conditions are favorable, rapid reproduction and accumulation can occur, and shellfish can accumulate the toxin to dangerous levels during such bloom events (Zingone and Enevoldsen 2000, Moore et al. 2009).

WDOH closes areas for shellfish harvest when PSP toxin levels equal or exceed the Food and Drug Administration standard of 80 micrograms (μg) of toxin per 100 grams of shellfish tissue. Areas are not reopened until testing has confirmed that the PSP toxin has declined to a safe level. Butter clams may experience extended closures because they typically retain the PSP toxin longer than other shellfish (up to one year).

Sentinel Mussel Monitoring Program

The Sentinel Mussel Monitoring Program is an early warning system for marine biotoxins established by WDOH. Mussels generally register PSP toxin levels more quickly than other shellfish. Consequently, mussels are used as "sentinels" to determine whether PSP toxins are increasing in a given area. Under this monitoring program, mussels are placed in cages and set in strategic growing areas throughout Puget Sound. Mussel samples are then collected either biweekly or monthly and tested for levels of PSP. Rising PSP levels in these mussels trigger more targeted and frequent sampling regimens in other shellfish species in the affected area.

With assistance from local health jurisdictions, local tribes, the Puget Sound Restoration Fund, and volunteers, WDOH maintained and monitored 69 collection sites in 2008 (WDOH 2009). In addition to the sentinel mussel locations, commercial mussels were routinely monitored at Westcott Bay in San Juan Island and at Penn Cove in Whidbey Island.

Amnesic Shellfish Poisoning

Domoic acid is a naturally-occurring toxin produced by a species of microscopic marine diatoms of the genus *Pseudo-nitzschia*. The human illness known as ASP or domoic acid poisoning is caused by eating fish, shellfish or crab containing the toxin. ASP can result in gastrointestinal and neurological disorders within 24-48 hours of toxic shellfish consumption by humans, and can be life-threatening. There is no antidote for domoic acid poisoning and cooking does not destroy the toxin.

The razor clam and Dungeness crab fisheries on the outer coast of Washington State have incurred losses due to occurrences of domoic acid over the past two decades. In the fall of 1991, domoic acid was first detected in razor clams off the coast of Washington and caused several mild cases of ASP (Horner and Postel 1993). This prompted WDOH to begin monitoring all major shellfish growing areas for domoic acid. Research shows that razor clams accumulate domoic acid in the edible tissue (foot, siphon, and mantle) and are slow to rid themselves of the toxin (Wekell et al. 1994) due to the presence of a high affinity glutamate binding protein (Trainer and Bill 2004). However, razor clams can continue to function in marine environments with high concentrations of domoic acid (Trainer and Bill 2004), resulting in extended closures of shellfish beds of the outer coast of Washington. In Dungeness crab, domoic acid primarily accumulates in the viscera. The level of domoic acid determined to be unsafe for human consumption is 20 parts per million (ppm) in molluscan shellfish and 30 ppm for Dungeness crab viscera. Dungeness crab harvest areas are closed when three out of six individual crab viscera equals or exceeds 30 ppm.

Within Puget Sound, the first occurrence of domoic acid was in blue mussels harvested in Kilisut Harbor in 2003 (Bill et al. 2006), raising concerns about the possibility of shellfish closures similar to those on the outer coast. Puget Sound was presumed to be less susceptible to domoic acid closures due to the absence of harvested species (razor clams and Dungeness crab) that retain domoic acid for long periods. Many shellfish species that are harvested in Puget Sound, such as mussels, littleneck clams, and oysters, are able to depurate domoic acid over a period of hours or days (Novaczek 1992), whereas the ability of other species such as geoduck to retain or release domoic acid has not yet been determined (Trainer et al. 2007).

Heterosigma

While not responsible for illnesses in humans, blooms of the small, unicellular, flagellated raphidophyte *Heterosigma akashiwo* have been shown to kill fish through the likely production of neurotoxins that disrupt respiratory and osmoregulatory gill functions (Khan et al. 1997, Hard et al. 2000). Farmed fish are particularly susceptible to mortality from increased concentrations of *Heterosigma* (Chang et al. 1990, Hard et al. 2000). Increased water column stratification and high temperatures have both been correlated with *Heterosigma* blooms although the precise causes for blooms remain uncertain (Bearon et al. 2006, O'Halloran et al. 2006).

Ulvoids

Blooms of ulvoid seaweeds are manifested by large quantities of green algal biomass washing up on beaches where decomposition occurs or in seagrass beds where mortality of seagrass through smothering is possible (den Hartog 1994, Nelson and Lee 2001). The thin blade-like morphology of ulvoids is thought to contribute to their ability to respond quickly to favorable environmental conditions such as increased nutrients and light (e.g., Littler and Littler 1980). As such, they have can competitively displace other algal species and seagrasses (e.g., den Hartog 1994, Anderson et al. 1996, Valiela et al. 1997). While not typically associated with the production of toxins, there is emerging evidence that ulvoid algae can produce allelopathic compounds that are detrimental to the development and growth of invertebrate larvae and other algae (Nelson et al. 2003a, Van Alstyne et al. 2007). Two genera of ulvoid seaweeds are common in Puget Sound: *Ulva* (which includes the former genus *Enteromorpha*) and *Ulvaria* (formerly referred to as *Monostroma*)(Nelson et al. 2003b). Despite their morphological similarity, these genera differ ecologically. A combination of field and lab observations conducted by Nelson and colleagues has demonstrated that *Ulva* is more tolerant of desiccation stress, produces lower levels of allelopathic compounds and is found more commonly in intertidal habitats whereas *Ulvaria* is less tolerant of desiccation stress, produces higher levels of allelopathic compounds and is more commonly found in subtidal habitats (Nelson et al. 2003a, Nelson et al. 2003b, Nelson et al. 2008, Nelson et al. 2010).

Status

PSP and ASP

In 2008, only 12 of 2,798 samples (0.4%) of shellfish tested by the Washington State Public Health Laboratory detected levels of PSP toxins greater than 1,000 micrograms and no PSP-related illnesses in humans were reported (WDOH 2009) (Table 1). However, 23 subtidal geoduck clamtracts were closed due to elevated PSP toxin levels and two general closures for "all shellfish species" occurred. Notably, one geoduck tract closure included a recall of 3,368 lbs of geoduck clams. In 2008, the highest PSP levels in blue mussels were found in Mystery Bay, Kilisut Harbor (Table 1).

Table 1. Areas of highest PSP levels in 2008 (WDOH 2009)

Date	Harvest Area	Species	Toxin Level*

08/10/2008	Mystery Bay, Kilisut Harbor	Blue Mussel	2,602
06/17/2008	Semiahmoo Marina, Drayton Harbor	Blue Mussel	1,831
08/11/2008	Scow Bay, Kilisut Harbor	Blue Mussel	1,779
09/25/2008	Dockton, Quartermaster Harbor	Blue Mussel	1,462
06/17/2008	Birch Bay Village, Birch Bay	Blue Mussel	1,456
08/06/2008	Fort Flagler, Kilisut Harbor	Blue Mussel	1,347
11/12/2008	Ediz Hook, East Straits	Blue Mussel	1,097

- micrograms per 100 grams of shellfish meat tissue

Approximately 12 Dungeness crab and 1,318 molluscan shellfish samples were tested by WDOH for domoic acid in 2008. The low sample size for crabs was driven by lack of toxin in the first 12 samples, which prompted a halt in further testing of Dungeness crab. There were no shellfish closures due to high levels of domoic acid in 2008, nor any reported ASP illnesses (WDOH 2009). The highest levels of domoic acid observed in Puget Sound molluscs in 2008 were at Squaxin Passage and Budd Inlet (Table 2).

Table 2. Areas of highest domoic acid levels in 2008 (WDOH 2009)

Date	Harvest Area	Species	Toxin Level*
06/24/2008	Squaxin Passage	Blue Mussel	3
06/19/2008	Budd Inlet	Blue Mussel	3
01/07/2008	Kalaloch Beach North	Razor Clam	3
11/06/2008	Long Beach Reserve	Razor Clam	2
10/01/2008	Sequim Bay	Blue Mussel	2
06/24/2008	South Tacoma Narrows	Blue Mussel	2

- parts per million per 1 gram of shellfish meat tissue

Heterosigma

Heterosigma has been reported in various locations in Puget Sound and has been linked to fish mortality at fish farms in Puget Sound (Hershberger et al. 1997, Tyrrell et al. 2002). Despite the potential problem of financial damage to fish farms from Heterosigma or of mortality of wild fish from *Heterosigma* blooms, available data the spatial variation of both *Heterosigma* occurrence and the frequency of associated fish mortality events is limited.

Ulvoids

A study conducted by Nelson et al. (2003b) assessing biomass of ulvoids in locations in Puget Sound (Figure 1) in summer of 2000 found that the species composition, depth, and abundance of ulvoids was variable throughout Puget Sound (Figure 2). In a more detailed analysis linking ulvoid biomass to abiotic variables on the coast of Blakely Island in the San Juan Archipelago, Nelson et al. (2003b) found that increased biomass was positively correlated with increased nitrogen, a finding that is consistent with studies conducted in other locations (e.g., Sfriso et al. 1992, Anderson et al. 1996).

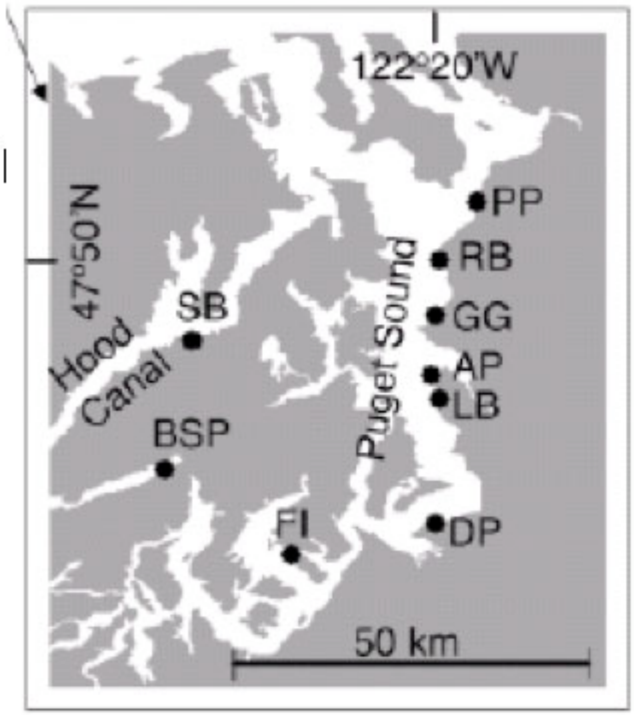


Figure 1. Sampling locations for ulvoid algae conducted by Nelson et al.(2003b)(Reprinted with permission from Botanica Marina and De Gruyter Publishing).

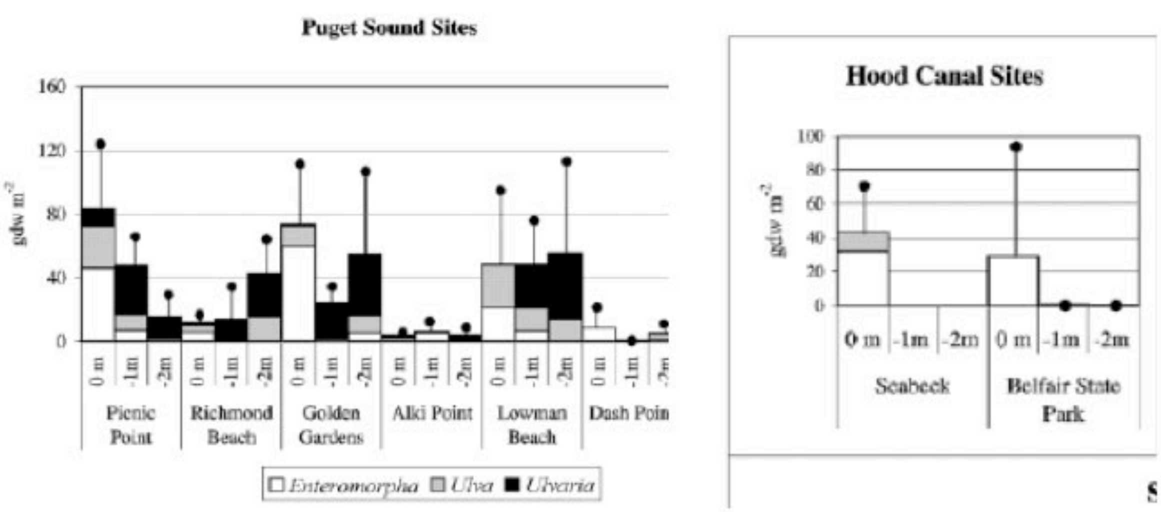


Figure 2. Biomass (mean + 1SD) by genus and by depth if ulvoid algae at locations throughout Puget Sound (Nelson et al. 2003b)(Reprinted with permission from Botanica Marina and De Gruyter Publishing).

Trends

PSP

Harmful algal blooms of *Alexandrium* were widespread and prevalent in the northern regions of Puget Sound (e.g., Sequim and Discovery Bays) in the 1950s and 1960s, but extended southward in the 1970s and 1980s to inner regions of Puget Sound (Trainer et al. 2003). More recent occurrences of PSP toxins in Washington shellfish and crab have been variable. Although high levels of PSP were detected in many years between 1990 and 2006, in some years (e.g., 1995, 2007, 2008) PSP toxin levels remained low (Figure 3). Despite this variability, the frequency of instances of high levels of PSP toxins detected by WDOH monitoring in Washington State has increased since 1957(Figure 3), a trend that is consistent with a worldwide increase in PSP toxic events since the 1950s (Nishitani and Chew 1984, Hallegraeff 1993, Trainer et al. 2003, Maso and Garces 2006).

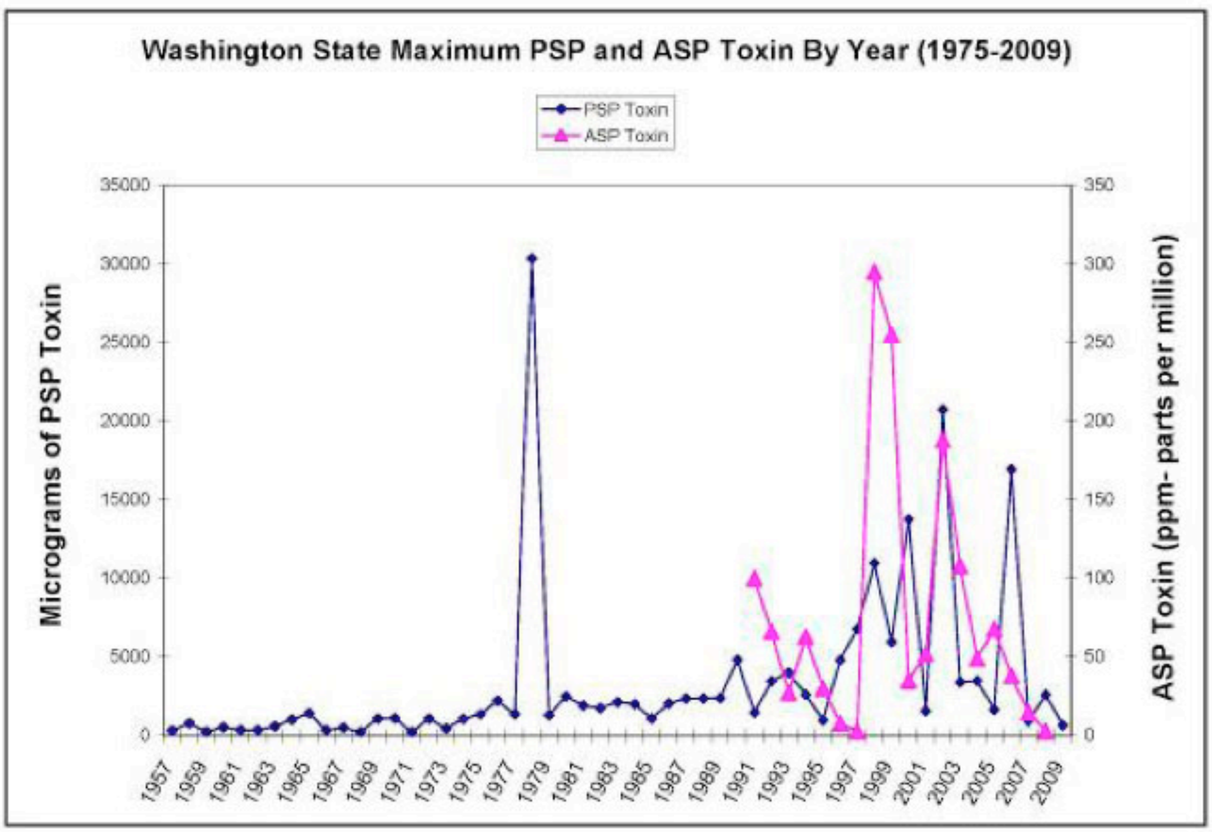


Figure 3. Annual maximum concentrations of PSP and ASP toxins observed in Washington State (WDOH, reprinted from Puget Sound Partnership 2009).

Moore et al. (2009) caution the use of PSP levels in shellfish tissues as a proxy for algal cell density in the water column due to the difference in accumulation and depuration rates of shellfish species. To investigate trends and possible relationships with large-scale climate and local environmental factors, Moore et al. (2009) analyzed PSP levels in blue mussels between 1993 and 2007. A combination of warm air and water temperatures and low streamflow appears to be favorable for PSP toxin accumulation in mussels, but advanced warning of events may be constrained by the same factors as for weather prediction, and is therefore limited to approximately one to two weeks (Moore et al. 2008, Moore et al. 2009). No increase in the frequency, magnitude, duration, or geographic scope of HAB events was detected, yet a significant basin-wide trend for closures to be imposed earlier in the year was observed over the period.

ASP

Blooms of *Pseudo-nitzchia* continue to affect Washington's outer coast since the first fisheries closure due to ASP toxins in 1991. Exceptional years of domoic acid-associated beach, razor clam, and Dungeness crab closures in Washington include 1991, 1998-1999, 2002-2003, and 2005 (Horner and Postel 1993, Trainer et al. 2007)(Figure 3). The prolonged closures of 1998-1999 and 2002-2003 (>1.5 years) resulted in significant commercial, recreational, and tribal shellfish harvest losses in Washington State (Dyson and Huppert in press, corrected proof). Out of concern for ASP toxins in the highly populated Puget Sound region, WDOH has monitored throughout Puget Sound since 1991. *Pseudo-nitzschia* blooms were reported in Puget Sound in 2003 and 2005, causing concern that blooms could impact the valuable fisheries there (Trainer et al. 2009).

Heterosigma

The bloom-forming alga *Heterosigma akashiwo* is recognized as a potential problem in Puget Sound. Despite a number of current studies on this HAB-forming alga, data are not yet available to determine spatial and temporal trends in *Heterosigma* abundances or the frequency of toxic events in Puget Sound.

Ulvoids

Published accounts of temporal trends in ulvoid abundances in Puget Sound are lacking. At least one investigation currently is underway to estimate ulvoid abundance from archival video surveys.

Uncertainties

Trend analysis of harmful algal blooms is difficult due to the lack of understanding about the dynamics that drive them, although this is an area of active research (e.g., Bearon et al. 2006, Nelson et al. 2008, Moore et al. 2009). Environmental conditions such as circulation, temperature, sunlight, nutrients, and salinity as well as the presence of algal predators, parasites and algal disease organisms all likely play a role in the formation, magnitude, and persistence of blooms. While PSP and ASP toxin levels currently are monitored and reported by WDOH, published data regarding spatial and temporal trends in *Heterosigma* and ulvoid abundances in Puget Sound are lacking.

Summary

Harmful algal blooms in Puget Sound have been variable over the past two decades, but appear to be increasing since WDOH began monitoring in 1957. Current monitoring efforts are not sufficient to provide accurate forecasting of ASP and PSP-related bloom events beyond one to two weeks, but forecasting could be improved by increased temporal and spatial scale and automated devices. While there is emerging concern about blooms of *Heterosigma* and ulvoids, , data that address these concerns currently is lacking for Puget Sound.

Literature Cited

Anderson, R. J., P. M. S. Monteiro, and G. J. Levitt. 1996. The effect of localised eutrophication on competition between *Ulva lactuca* (Ulvaceae, Chlorophyta) and a commercial resource of *Gracilaria verrucosa* (Gracilariaceae, Rhodophyta). Hydrobiologia 327:291-296.

Bearon, R. N., D. Grunbaum, and R. A. Cattolico. 2006. Effects of salinity structure on swimming behavior and harmful algal bloom formation in *Heterosigma akashiwo*, a toxic raphidophyte. Marine Ecology-Progress Series 306:153-163.

Bill, B. D., F. H. Cox, R. A. Horner, J. A. Borchert, and V. L. Trainer. 2006. The first closure of shellfish harvesting due to domoic acid in Puget Sound, Washington, USA. African Journal of Marine Science 28:435-440.

Chang, F. H., C. Anderson, and N. C. Boustead. 1990. 1st Record of a *Heterosigma* (Raphidophyceae) bloom with associated mortality of cage-reared salmon in Big Glory Bay, New Zealand New Zealand Journal of Marine and Freshwater Research 24:461-469.

den Hartog, C. 1994. Suffocation of a littoral *Zostera* bed by *Enteromorpha radiata*. Aquatic Botany 47:21-28.

Determan, T. A., S. Washington . Office of Food, P. Shellfish, and P. Puget Sound Ambient Monitoring. 2001. Paralytic shellfish poisoning (PSP) patterns in Puget Sound shellfish, year 2000 : a report for the Puget Sound Ambient Monitoring Program. Office of Food Safety and Shellfish Programs, Washington State Dept. of Health, [Olympia, Wash.].

Dyson, K., and D. D. Huppert. in press, corrected proof. Regional economic impacts of razor clam beach closures due to harmful algal blooms (HABs) on the Pacific coast of Washington. Harmful Algae In Press, Corrected Proof.

Hallegraeff, G. 1993. A review of harmful algae blooms and their apparent global increase. Phycologia 32:79 - 99.

Hard, J. J., L. Connell, W. K. Hershberger, and L. W. Harrell. 2000. Genetic variation in mortality of chinook salmon during a bloom of the marine alga *Heterosigma akaskiwo*. Journal of Fish Biology 56:1387-1397.

Hershberger, P. K., J. E. Rensel, A. L. Matter, and F. B. Taub. 1997. Vertical distribution of the chloromonad flagellate *Heterosigma carterae* in columns: implications for bloom development. Canadian Journal of Fisheries and Aquatic Sciences 54:2228-2234.

Horner, R. A., and J. R. Postel. 1993. Toxic diatoms in western Washington waters (US West coast). Hydrobiologia 269:197-205.

Khan, S., O. Arakawa, and Y. Onoue. 1997. Neurotoxins in a toxic red tide of *Heterosigma akashiwo* (Raphidophyceae) in Kagoshima Bay, Japan. Aquaculture Research 28:9-14.

Littler, M. M., and D. S. Littler. 1980. The evolution of thallus form and survival strategies in benthic marine macroalgae: Field and laboratory tests of a functional form model. The American Naturalist 116:25.

Maso, M., and E. Garces. 2006. Harmful microalgae blooms (HAB); problematic and conditions that induce them. Marine Pollution Bulletin 53:620-630.

Moore, S., V. Trainer, N. Mantua, M. Parker, E. Laws, L. Backer, and L. Fleming. 2008. Impacts of climate variability and future climate change on harmful algal blooms and human health. Environmental Health 7:S4.

Moore, S. K., N. J. Mantua, B. M. Hickey, and V. L. Trainer. 2009. Recent trends in paralytic shellfish toxins in Puget Sound, relationships to climate, and capacity for prediction of toxic events. Harmful Algae 8:463-477.

Nelson, T. A., K. Haberlin, A. V. Nelson, H. Ribarich, R. Hotchkiss, K. L. V. Alstyne, L. Buckingham, D. J. Simunds, and K. Fredrickson. 2008. Ecological and physiological controls of species composition in green macroalgal blooms. Ecology 89:1287-1298.

Nelson, T. A., and A. Lee. 2001. A manipulative experiment demonstrates that blooms of the macroalga *Ulvaria obscura* can reduce eelgrass shoot density. Aquatic Botany 71:149-154.

Nelson, T. A., D. J. Lee, and B. C. Smith. 2003a. Are "green tides" harmful algal blooms? Toxic properties of water-soluble extracts from two bloom-forming macroalgae, *Ulva fenestrata* and *Ulvaria obscura* (Ulvophyceae). Journal of Phycology 39:874-879.

Nelson, T. A., A. V. Nelson, and M. Tjoelker. 2003b. Seasonal and spatial patterns of "Green tides" (Ulvoid algal blooms) and related water quality parameters in the coastal waters of Washington state, USA. Botanica Marina 46:263-275. De Gruyter (Publisher).

Nelson, T. A., J. Olson, L. Imhoff, and A. V. Nelson. 2010. Aerial exposure and desiccation tolerances are correlated to species composition in "green tides" of the Salish Sea (northeastern Pacific). Botanica Marina 53:103-111.

Nishitani, L., and K. K. Chew. 1984. Recent developments in paralytic shellfish poisoning research. Aquaculture 39:317-329.

Novaczek, I., Madhyastha, M.S., Ablett, R.F., Donald, A., Johnson, G., Nijjar, M.S., Sims, D.E. 1992. Depuration of domoic acid from live blue mussels (*Mytilus edulis*). Can. J. Fish. Aquat. Sci. 49:312-318.

O'Halloran, C., M. W. Silver, T. R. Holman, and C. A. Scholin. 2006. *Heterosigma akashiwo* in central California waters. Harmful Algae 5:124-132.

Puget Sound Partnership. 2009. Ecosystem Status & Trends. Puget Sound Partnership, Seattle, WA.

Sfriso, A., B. Pavoni, A. Marcomini, and A. Orio. 1992. Macroalgae, nutrient cycles, and pollutants in the Lagoon of Venice. Estuaries and Coasts 15:517-528.

Trainer, V., B.-T. Eberhart, J. Wekell, N. Adams, L. Hanson, F. Cox, and J. Dowell. 2003. Paralytic shellfish toxins in Puget Sound, Washington State. J Shellfish Res 22:213 - 223.

Trainer, V. L., and B. D. Bill. 2004. Characterization of a domoic acid binding site from Pacific razor clam. Aquatic Toxicology 69:125-132.

Trainer, V. L., W. P. Cochlan, A. Erickson, B. D. Bill, F. H. Cox, J. A. Borchert, and K. A. Lefebvre. 2007. Recent domoic acid closures of shellfish harvest areas in Washington State inland waterways. Harmful Algae 6:449-459.

Tyrrell, J. V., L. B. Connell, and C. A. Scholin. 2002. Monitoring for *Heterosigma akashiwo* using a sandwich hybridization assay. Harmful Algae 1:205-214.

Valiela, I., J. McClelland, J. Hauxwell, P. J. Behr, D. Hersh, and K. Foreman. 1997. Macroalgal blooms in shallow estuaries: Controls and ecophysiological and ecosystem cnsequences. Limnology and Oceanography 42:1105-1118.

Van Alstyne, K., L. Koellermeier, and T. Nelson. 2007. Spatial variation in dimethylsulfoniopropionate (DMSP) production in *Ulva lactuca* (Chlorophyta) from the Northeast Pacific. Marine Biology 150:1127-1135.

WDOH, W. D. o. H. 2009. 2008 Annual Report: Commercial and Recreational Shellfish Areas in Washington State. Office of Shellfish and Water Protection.

Wekell, J. C., J. R. Gauglitz, E.J.Barnett, H.J.Hatfield, C.L.Simons, and D. Ayres. 1994. Occurrence of domoic acid in Washington state razor clams (*Siliqua patula*) during 1991–1993. Natural Toxins 2:197-205.

Zingone, A., and H. O. Enevoldsen. 2000. The diversity of harmful algal blooms: a challenge for science and management. Ocean & Coastal Management 43:725-748.

Dissolved Oxygen (Hypoxia)

Background

Hypoxia, defined as dissolved oxygen (DO) concentrations less than 2 mg / L, has become widespread throughout estuaries and semi-enclosed seas throughout the world (Diaz 2001). While hypoxia may be permanent or intermittent, it is most commonly manifested as a seasonal disturbance, appearing in mid- to late summer after vertical stratification prevents replenishment of deep water DO. The duration, extent and magnitude of seasonal hypoxia has dramatically increased over the past few decades in response to anthropogenic eutrophication (Diaz and Rosenberg 2008) and is now a common and regular feature in marine ecosystems that have strong vertical stratification and low flushing rates. Additionally, climate change may be altering the frequency and intensity of hypoxic conditions in coastal ecosystems (Chan et al. 2008).

Hypoxia is an important concern because low dissolved oxygen can have direct and indirect effects on marine communities and natural resources. Hypoxia and anoxia can be lethal to animals when oxygen levels are depleted beyond species physiological tolerances. For sessile organisms who have limited capacity to seek out refuges from hypoxia, direct lethal impacts may be most severe (Diaz and Rosenberg 1995). Mobile species generally act to minimize exposure to low DO through distributional shifts to refuges that have higher DO levels. While these responses minimize direct lethal impacts of low DO, they can induce indirect ecological effects such as reduced feeding rates, enhanced vulnerability to predators and reduced growth rates (Breitburg 1992, Breitburg et al. 1997, Eby and Crowder 2002, Bell et al. 2003, Craig and Crowder 2005, Aumann et al. 2006).

Status of hypoxia in Puget Sound

In many regions of Puget Sound, low DO is a natural consequence of its deep fjord-like bathymetry, where the water column stratification and slow flushing rates lead to long residence times of deep water that is not in contact with the atmosphere. However, there is some evidence that DO levels were generally higher in the mid 20th century than they are today (Newton et al. 1995). This conclusion is based on a comparison of historical water quality sampling data to contemporary data that used comparable techniques (Figure 1). Changes in the intensity of low DO conditions over a time period of increased human activity suggests a role for anthropogenic activity in dictating hypoxia.

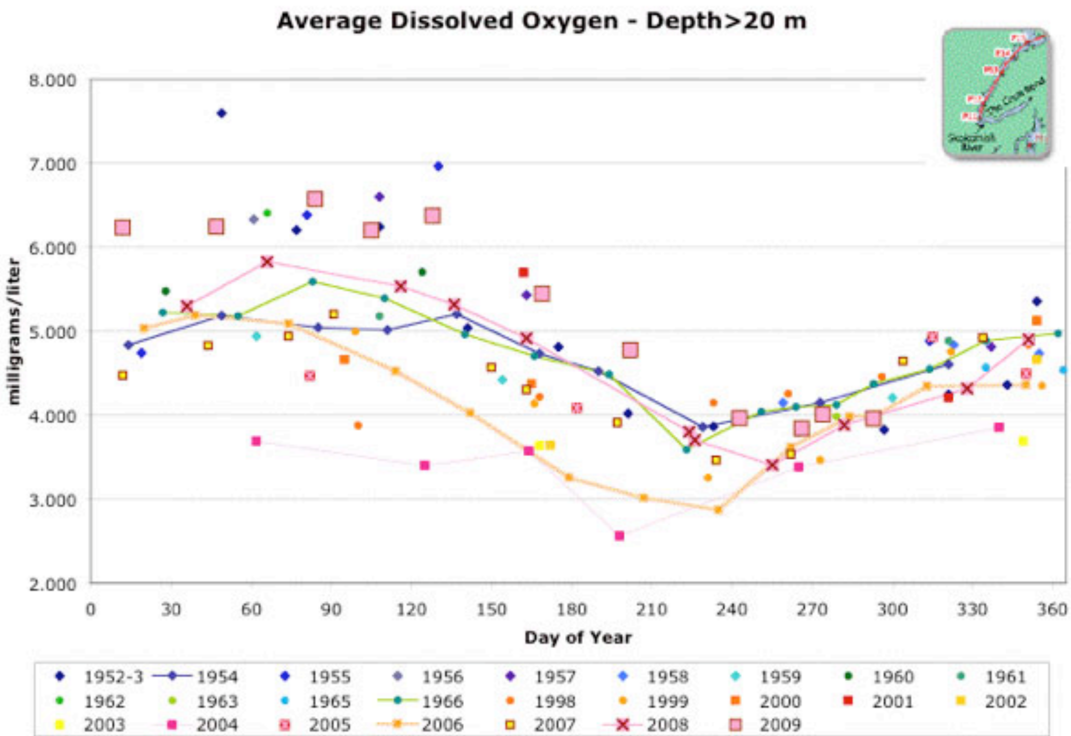


Figure 1. Integrated sub-surface water DO vs. day by sampling year in southern Hood Canal. Recent years with very low DO conditions (e.g., 2004, 2006) have no historical precedent over the period of record (1952 -1966). Data and analysis from Hood Canal Dissolved Oxygen Program: http://www.hoodcanal.washington.edu/. Figure produced by and used with permission from M.J. Warner, University of Washington.

Low dissolved oxygen is present seasonally in Puget Sound at several locations (Figure 2). Much of the southern one-half of Hood Canal now regularly experiences hypoxic conditions in mid- to late summer. Several regions within the south basin of Puget Sound are also prone to hypoxia (Albertson et al. 2007), especially Budd, Carr and Case Inlet (Albertson et al. 2002). Saratoga passage also is susceptible to low DO (Figure 2)(Albertson et al. 2002).

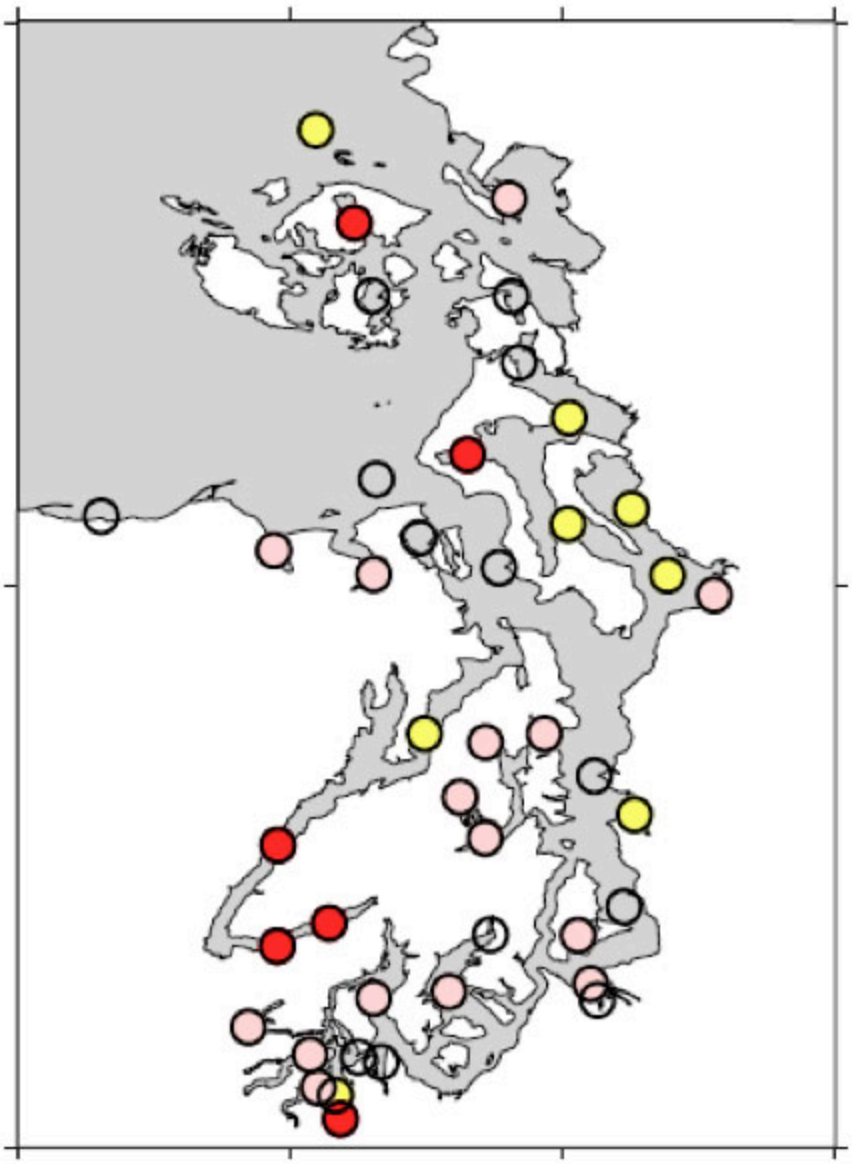


Figure 2. Washington State Department of Ecology water quality monitoring stations showing low (red, < 3 mg /l), stressful (yellow, <5 mg /l) and high (empty circles) DO, 1990-1997. Pink circles indicate stations likely to be to have low DO based on physico-chemical characteristics. Reprinted from Albertson et al. (2002) with permission from Washington Department of Ecology.

Since the mid-2000's, there has been a proliferation of monitoring efforts and web-based distribution of data, especially for Hood Canal. These include (1) monitoring via citizens that provides weekly along 6 stations that transverse Hood Canal (2) deploying of remove ocean observing systems (Oceanic Remote Chemical-optical Analyzer, ORCA) that provide high frequency monitoring of water conditions (3) routine monitoring via WA Department of Ecology.

These data can be downloaded from http://www.hoodcanal.washington.edu/. The expansion of data collection capacity has revealed the importance of oceanographic processes for determining the spatial patterning and temporal persistence of low DO in Hood Canal (Figure 3). In both Hood Canal and South Puget Sound, research activities are presently underway to develop high resolution models to predict DO levels and their sensitivity to surface flows and oceanographic conditions (Albertson et al. 2007). The below summary emphasizes insights gleaned from the study of Hood Canal, only because of the greater concentration of research activity in this basin.

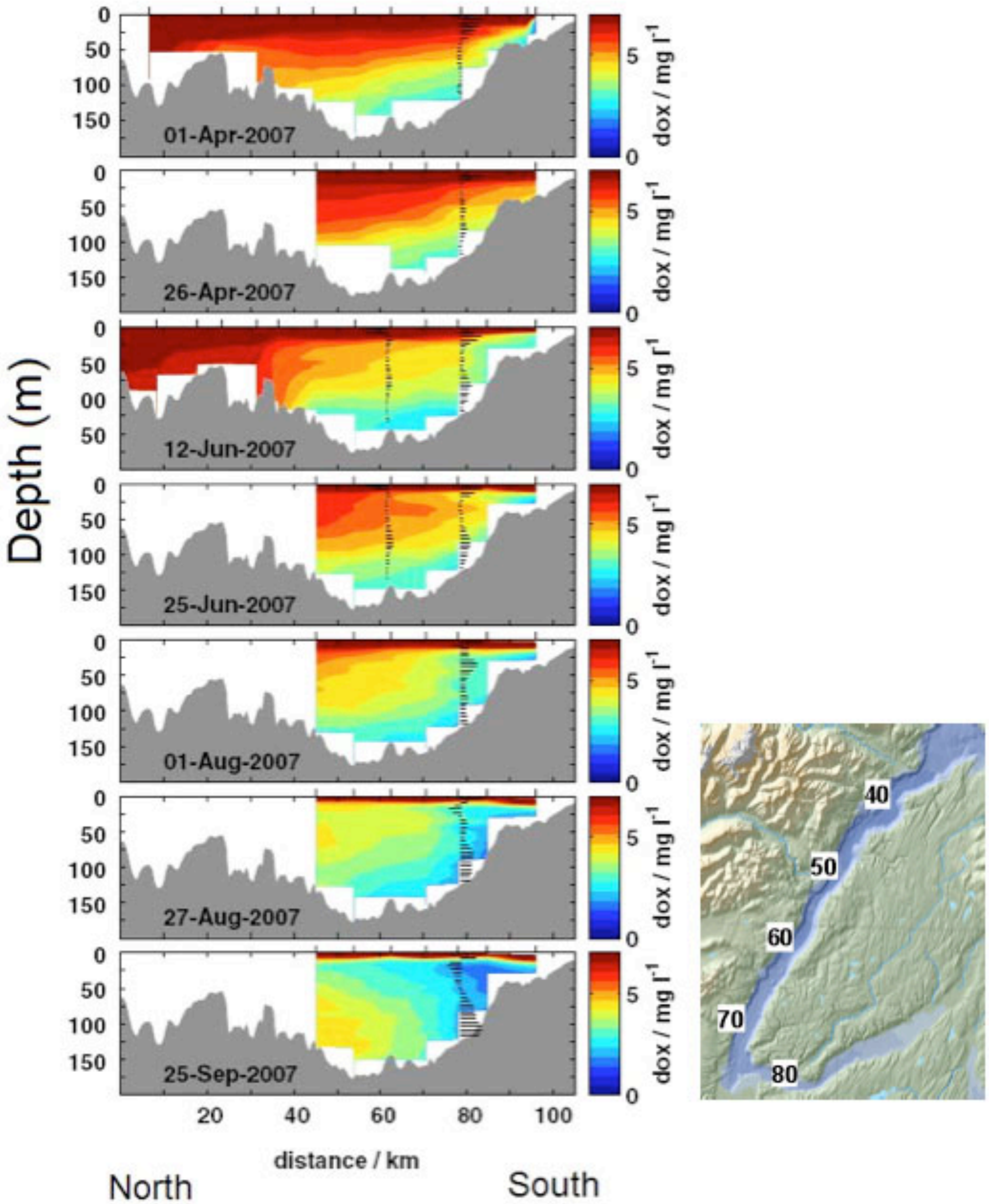


Figure 3. Cross sectional profiles of dissolved oxygen (DO) in Hood Canal, April – September 2007. Hypoxia first emerges at depth in the southern end of Hood Canal, and extends northward along the bottom until mid summer. In mid summer, the horizontal extent of hypoxia constricts southward but expands vertically through late summer / early autumn. Black bars indicate mean water velocities and direction. In the proposed study, the southern impact region spans roughly kilometers 70 – 80, while the north unimpacted region extends from 40 – 50. Inset map shows location of cross section distance markers (kilometers). Figure produced by Mickett and M.J. Warner and used with permission from M.J. Warner, University of Washington. http://www.hoodcanal.washington.edu/observations/ccross.jsp.

Anthropogenic Influences

Hypoxia is a symptom of eutrophication whereby excessive primary production fuels high rates of microbial respiration as sinking organic matter is decomposed in deep waters. Cultural eutrophication is caused by anthropogenic loading of nutrients that limited phytoplankton growth; in Puget Sound, dissolved inorganic nitrogen (DIN) is the primary limiting nutrient for primary producers (Newton et al. 1998). Thus, human activities that increase DIN loading directly promote hypoxic conditions. DIN enters the Puget Sound through multiple sources: (1) via the surrounding watershed via surface flow, groundwater, wastewater, and shallow septic systems (2) from recycling of nutrient from the sediments into the water column; (3) directly from the atmosphere and (4) from water exchange with the marine environment. Human activity primarily affects watershed-based inputs, although climate change could alter delivery of nitrogen from coastal marine waters.

Three primary anthropogenic activities are thought to be important in changing low DO conditions via DIN inputs into Puget Sound. The first is through the conversion of riparian vegetation from a community dominated by firs and cedars to one replaced with red alders (Busse and Gunkel 2001). As alders host symbiotic microorganisms that have the capacity to fix atmospheric nitrogen into a biologically available form, their current abundance may lead to increased nitrogen loading. The second is through shallow shoreline septic systems. A mass balance estimation of DIN loads to Hood Canal revealed that shallow ground water flow from shoreline septic systems contributed less than 5% of the total dissolved inorganic nitrogen to the upper water layer (Paulson et al. 2007). The third is from wastewater treatment plant discharge. In South Puget Sound, wastewater treatment comprises roughly one-half of the watershed-derived DIN loading (Roberts et al. 2008), but this component may be larger if water exchange with the central Basin is considered (Figure 4)(Roberts et al. 2008). At present, there are no published reports or papers that definitively implicate any single source as the most important cause of reduced DO.

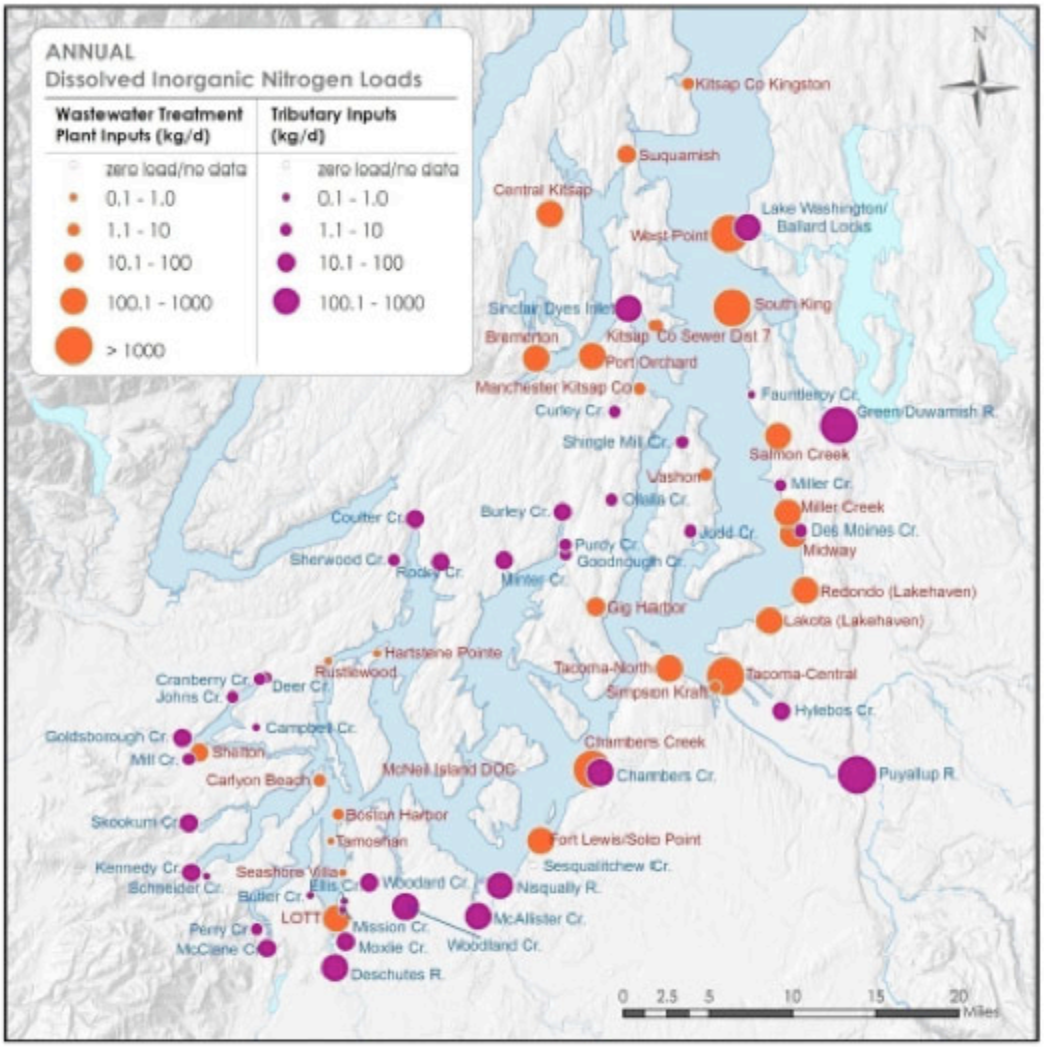


Figure 4. Annual DIN loads from freshwater surface flows and wastewater treatment plants. Reprinted from Roberts et al. (2008) with permission from Washington Department of Ecology.

Impacts on Biota

Hypoxia has been implicated in shifts in benthic infauna and in the pelagic community. Benthic infauna provide the most important source of food for most of the groundfish in Puget Sound so changes is these communities may have important ecological consequences. Long et al. (2007) demonstrated substantial shifts in community structure associated with water column dissolved oxygen levels below 3 mg/L. In general, the overall density of benthic infauna and species richness were reduced as dissolved oxygen decreased. Valero et al (2008) compared population dynamics of geoduck clams in southern and northern reaches of Hood Canal and implicated hypoxia as a significant factor in population declines in the southern region. Parker-Stetter and Horne (2009) described shifts in the distribution of pelagic organisms during a period of pronounced midwater anoxic zone during 2006, suggesting that severe midwater minimum layers can create a predation refuge for zooplankton. However, in the following year midwater oxygen minimum layers did not appear to affect the vertical distribution of fish and invertebrates, although it did appear to impact the rate of diel migration (Parker-Stetter et al. 2009). Palsson et al. (2008) described substantial vertical distributional shifts of rocky-reef associated fish species in response to the low dissolved oxygen event, but also noted that responses varied by species.

Several fish kill events in southern Hood Canal have been documented (2002, 2003, 2006), all occurring in late summer. Fish kill events correspond with a rapid vertical displacement of hypoxic / anoxic water, such that mobile fishes are unable to mount behavioral responses quickly enough to avoid exposure. The 2003 and 2006 fish kill events were differentiated by the primary species affect: copper rockfish were the dominant species affected in the 2003 event, while lingcod were more impacted by the 2006 event (Palsson et al. 2008). The ratios of dead : total observed copper rockfish ranged from 0 – 26%, while for lingcod these rations ranged from 3 – 37% (Palsson et al. 2008)

CONTENT PENDING REVIEW

Added: 10/7/2010

Author: Dr. Tim Essington, School of Aquatic and Fisheries Science, University of Washington

Essington and Paulsen (2010) used a comparative approach to ask whether there was evidence of hypoxia on densities of demersal fish and macroinvertebrates in southern Hood Canal. They found strong evidence supporting the hypothesis that sessile macroinvertebrate densities are strongly reduced by hypoxia: the five main species sampled were generally reduced in abundance by ca. 90% compared to what would be expected based from the reference sites. In contrast, there was little evidence for persistent density reductions in mobile fauna. However, mobile macroinvertebrates and fishes exhibited significant density reductions in southern Hood Canal during late summer when hypoxia was present, presumably due to behavioral distributional responses that displaced individuals from southern Hood Canal. The large reduction in demersal species' densities suggests substantial effects of hypoxia in Hood Canal even at oxygen levels that were marginally hypoxic (2 mg / l). They conclude that understanding the full ecological consequence of hypoxia will require a greater knowledge on the spatial extent of distributional shifts and their effects on competitive and predator–prey interactions.

Reference:

Essington, TE and Paulsen, CE 2010. Quantifying hypoxia impacts on an estuarine demersal community using a hierarchical ensemble approach. Ecosystems. 13: published on-line prior to print doi:10.1007/s10021-010-9372-z

Uncertainties

Identifying the ultimate causes of hypoxia and policy responses that might mitigate them remains a high priority. Because of high interannual variability, it is not possible to discern whether the intensity or spatial extent of hypoxia has been growing over recent years. Moreover, the long-term effects of regular exposure to seasonal hypoxia on communities and food webs has not yet been published. Valuable species such as geoduck clams and Dungeness crabs may be adversely affected by hypoxic conditions, though it is not yet possible to definitively quantify the contribution of hypoxia to putative population declines in hypoxia-impacted regions.

Literature Cited

Albertson, S. L., J. Bos, G. Pelletier, and M. Roberts. 2007. Estuarine flow int he south basin of Puget Sound and its effects on near-bottom dissolved oxygen. WA Department of Ecology, Olympia, WA.

Albertson, S. L., K. Erickson, J. A. Newton, G. Pelletier, R. A. Reynolds, and M. Roberts. 2002. South Puget Sound Water Quality Study Phase 1. WA Department of Ecology, Olympia, WA.

Aumann, C. A., L. A. Eby, and W. F. Fagan. 2006. How transient patches affect population dynamics: The case of hypoxia and blue crabs. Ecological Monographs 76:415-438.

Bell, G. W., D. B. Eggleston, and T. G. Wolcott. 2003. Behavioral responses of free-ranging blue crabs to episodic hypoxia. II. Feeding. Marine Ecology-Progress Series 259:227-235.

Breitburg, D. 1992. Episodic hypoxia in Chesapeake Bay: Interacting effects of recruitment, behavior, and physical disturbance. Ecological Monographs 62:525-546.

Breitburg, D. L., T. Loher, C. A. Pacey, and A. Gerstein. 1997. Varying effects of low dissolved oxygen on trophic interactions in an estuarine food web. Ecological Monographs 67:489-507.

Busse, L. B. and G. Gunkel. 2001. Riparian alder fens - Source or sink for nutrients and dissolved organic carbon?:1. Effects of water level fluctuations. Limnologica 31:307-315.

Chan, F., J. A. Barth, J. Lubchenco, A. Kirinich, H. Weeks, W. T. Peterson, and B. A. Menge. 2008. Emergence of anoxia in the California current large marine ecosystem. Science 319:920.

Craig, J. K. and L. B. Crowder. 2005. Hypoxia-induced habitat shifts and energetic consequences in Atlantic croaker and brown shrimp on the Gulf of Mexico shelf. Marine Ecology-Progress Series 294:79-94.

Diaz, R. J. 2001. Overview of hypoxia around the world. Journal of Environmental Quality 30:275-281.

Diaz, R. J. and R. Rosenberg. 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. : Oceangraphy and Marine Biology : an Annual Review 33:245-303.

Diaz, R. J. and R. Rosenberg. 2008. Spreading dead zones and consequences for marine ecosystems. Science 321:926-929.

Eby, L. A. and L. B. Crowder. 2002. Hypoxia-based habitat compression in the Neuse River Estuary: context-dependent shifts in behavioral avoidance thresholds. Canadian Journal Of Fisheries And Aquatic Sciences 59:952-965.

Long, E. R., M. E. Dutch, S. Aasen, K. I. Welch, V. A. Partridge, and D. A. Shull. 2007. Relationships between the composition of the benthos and sediment and water quality parameters in Hood Canal: Task IV Hood Canal Dissolved Oxygen Program. WA Department of Ecology, Olympia, WA.

Newton, J. A., M. Edie, and J. Summers. 1998. Primary productivity in Budd Inlet during 1997: Seasonal patterns of variation and controlling factors.in Puget Sound Research 1998.

Newton, J. A., A. L. Thomson, L. B. Eisner, G. A. Hannach, and S. L. Albertson. 1995. Dissolved oxygen concentrations in Hood Canal: are conditions different than forty years ago?in Puget Sound Research '95.

Palsson, W. A., R. E. Pacunski, T. R. Parra, and Beam.J. 2008. The effects of hypoxia on marine fish populations in Southern Hood Canal, Washington. American Fisheries Society Symposium 64:255-280.

Parker-Stetter, J. L. and J. K. Horne. 2009. Nekton distribution and midwater hypoxia: a seasonal, diel prey refuge? Estuaries and Coasts 81:13-18.

Parker-Stetter, S. L., J. K. Horne, and M. M. Langness. 2009. The influence of midwater hypoxia on nekton vertical migration. ICES Journal of Marine Science 66:1296-1302.

Paulson, A. J., C. P. Konrad, L. M. Frans, M. Noble, C. Kendall, E. G. Josberger, R. L. Huffman, and T. D. Olsen. 2007. Freshwater and saline loads of dissolved inorganic nitrogen to Hood Cana and Lynch Cove, Washington. U.S. Gelogical Survey, Reston, VA.

Roberts, M., J. Bos, and S. L. Albertson. 2008. South Puget Sound dissolved oxygen study interim data report. WA Department of Ecology, Olympia, WA.

Valero, J. L., B. Lee, D. Armstrong, L. Orensanz, A. Parma, R. Hilborn, B. Sizemore, and T. Palzer. 2008. Population dynamics and historic trends of geoduck clams under episodic low dissolve oxygen conditions in Hood Canal. Journal Of Shellfish Research 27:462-463.

Eutrophication of Marine Waters

Background

Eutrophication of water bodies occurs when high levels of nutrients fuel high rates of primary production and accumulation of algal biomass, either as macroalgae or phytoplankton. Some ecosystems are naturally eutrophic, but in others human activity causes ecosystems to undergo transformations into a eutrophic state. This is termed cultural eutrophication, and is the primary concern in evaluating the status of marine waters of Puget Sound.

The primary cause of cultural eutrophication is human actions (e.g., land use, wastewater, agriculture) that increase the loadings of nutrients that limit algal growth (Carpenter et al. 1998). In Puget Sound (like many estuaries), dissolved inorganic nitrogen (DIN) is the primary limiting nutrient (Newton and Van Voorhis 2002). Research efforts have therefore focused on measuring the availability of DIN and on the rates of delivery from alternative sources. In general, DIN in Puget Sound can come from (1) the surrounding watershed via surface flow, groundwater, wastewater, and shallow septic systems; (2) recycling of nutrients from the sediments into the water column; (3) directly from the atmosphere; and (4) exchange with the coastal ocean. Human activity primarily affects watershed-based inputs, although climate change could alter delivery of nitrogen from coastal marine waters through its effects on coastal upwelling.

The vulnerability of an ecosystem to cultural eutrophication depends on several factors. Generally, strong vertical mixing can act to reduce the effects of nutrient enrichment via inducing light limitation on planktonic producers. Many areas of Puget Sound experience regular mixing through tidal exchange processes that could act to reduce the effects of anthropogenic DIN loading (Figure 1), but some are less well mixed and are therefore vulnerable to eutrophication. Such areas tend to be inlets with few freshwater inputs, and deep fjord-like basins that have limited exchange with surrounding waters (e.g., Hood Canal, South Puget Sound; Figure 1). A second major consideration is the extent to which primary production is already limited by DIN. This depends in large part on the availability of N from other sources: if DIN supply from other sources is relatively large, impacts of smaller additions of total N from anthropogenic sources may be relatively small. In Puget Sound, much of the DIN derives from exchange with coastal marine waters via exchanges in the Strait of Juan de Fuca and subsequently in the major sub-basins of Puget Sound (Mackas and Harrison 1997). A final consideration is the residence time of surface waters: if systems are rapidly flushed then surface waters containing anthropogenic DIN will be displaced quickly.

Figure 1. Sampling stations containing strong and persistent vertical stratification (red), based on WA Department of Ecology and PRISM data. Sites denoted by yellow and green are at lower risk of eutrophication. Reprinted from U.S. E.P.A. Region 10 Puget Sound Georgia Basin Ecosystem Indicators (for supporting references, see U.S. Environmental Protection Agency (2006).

In Puget Sound, the extent of DIN- limitation on algae varies strongly with space and time (Newton and Van Voorhis 2002) (Figure 2). In general, response of phytoplankton to nutrient enrichment is greatest during May – Aug. Nutrient responses in these months correspond to a drawing down of available DIN in the surface mixed layer during the spring, when phytoplankton production and standing stocks are the greatest (Newton and Van Voorhis 2002, Stark et al. 2008) (Figure 3a, 3b).

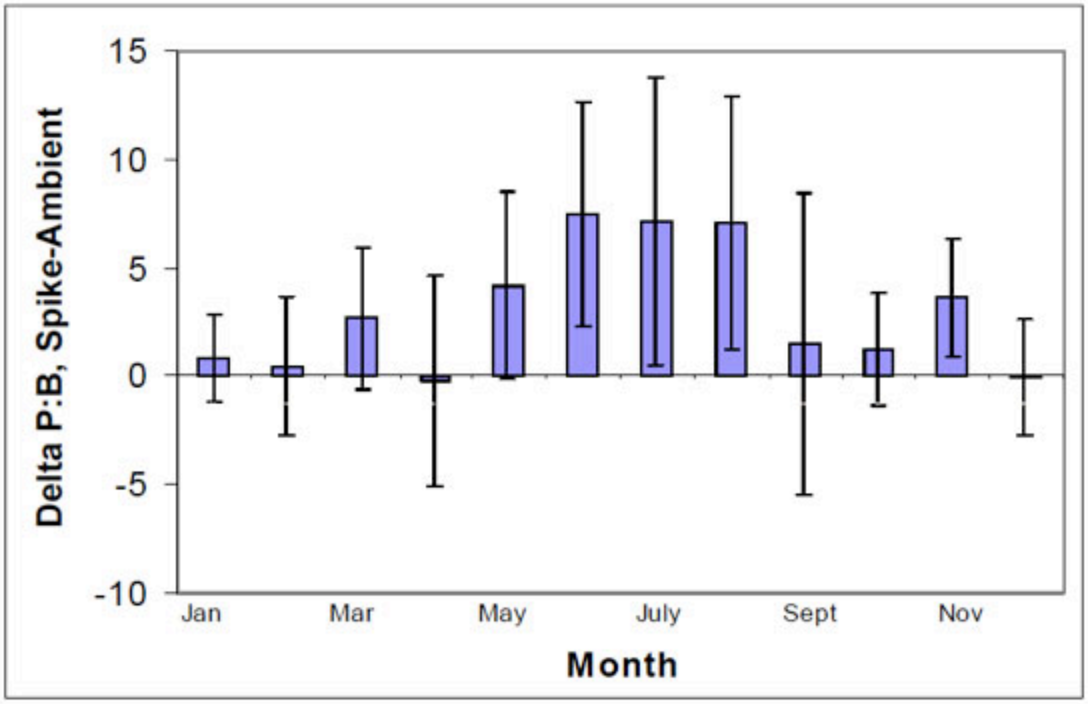


Figure 2. Change in phytoplankton production (production : biomass; PB) in response to nutrient spike. Bars represent averages taken over multiple sites. Nutrient limitation is greatest in May - August. Reprinted from Newton and Van Voorhis (2002) with permission from Washington Department of Ecology.

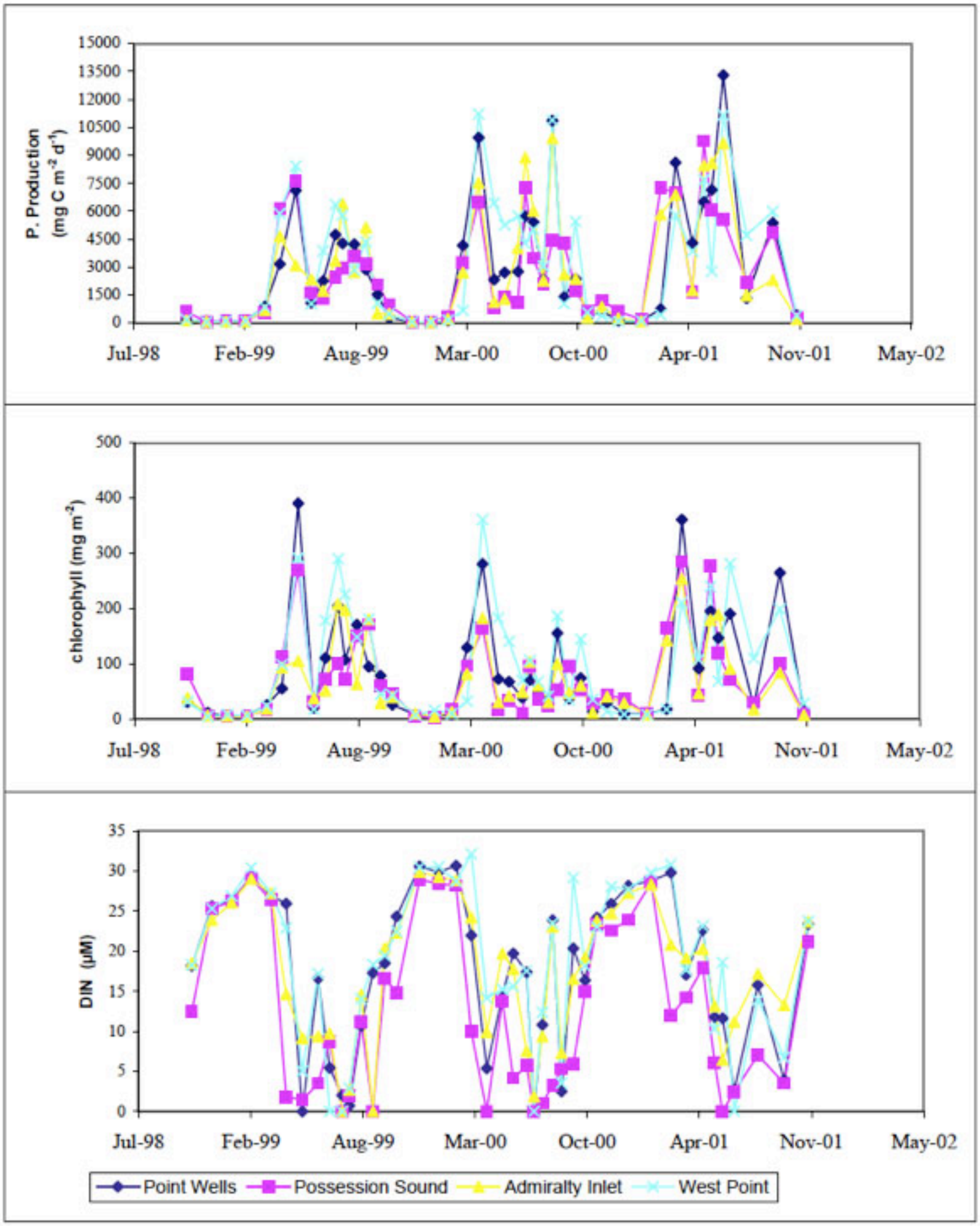


Figure 3a. Seasonal patterns of primary productivity, Chl.A and DIN at four sites, 1998 -2001. Reprinted from Newton and Van Voorhis (2002) with permission from Washington Department of Ecology.

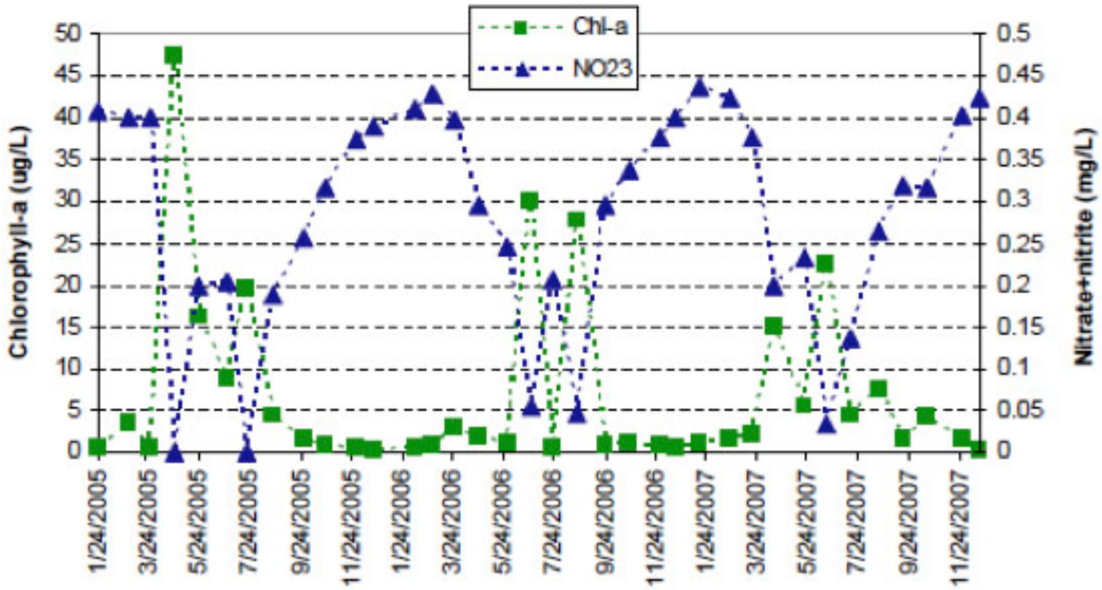


Figure 3b. Seasonal patterns of chlorophyll level and Nitrate/ Nitrite at Point Wells monitoring station, 2005-2007. Phytoplankton blooms are associated with a draw down of available DIN. Reprinted from Stark et al. (2008) with permission from King County Department of Natural Resources and Parks.

Monitoring Programs

Several entities conduct regular water quality monitoring within Puget Sound. The Washington State Department of Ecology conducts monthly sampling at several sites throughout Puget Sound (Figure 4). King County conducts monthly sampling at 14 offshore stations and 18 beach / nearshore stations in Central Puget Sound. The University of Washington PRISM program conducts biannual sampling at 39 stations throughout Puget Sound (Figure 5). The Hood Canal Dissolved Oxygen Program maintains 4 moorings that provide high-frequency monitoring of water quality conditions, and King County maintains three active moorings in central Puget Sound. Although the design of some of these monitoring programs have evolved over time to adapt to emerging issues, core sites have been maintained so that long-term trends can be evaluated(Newton et al. 2002). Detailed QA/ QC procedures for many of these programs are well documented (Washington State Department of Ecology 2006b, Albertson et al. 2007a).

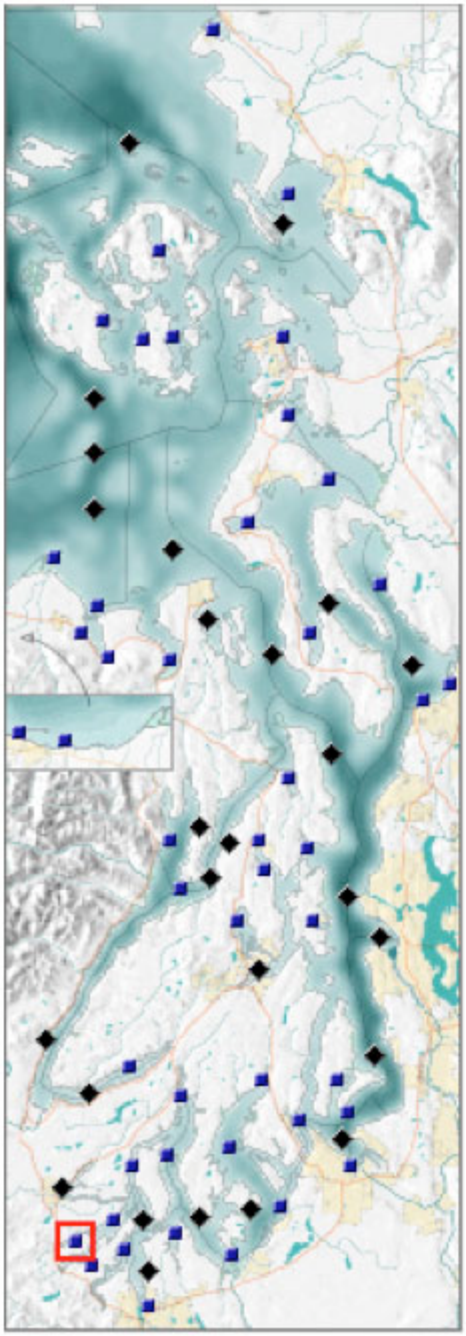


Figure 4. Location of Department of Ecology sampling sites. Used with permission from Department of Ecology.

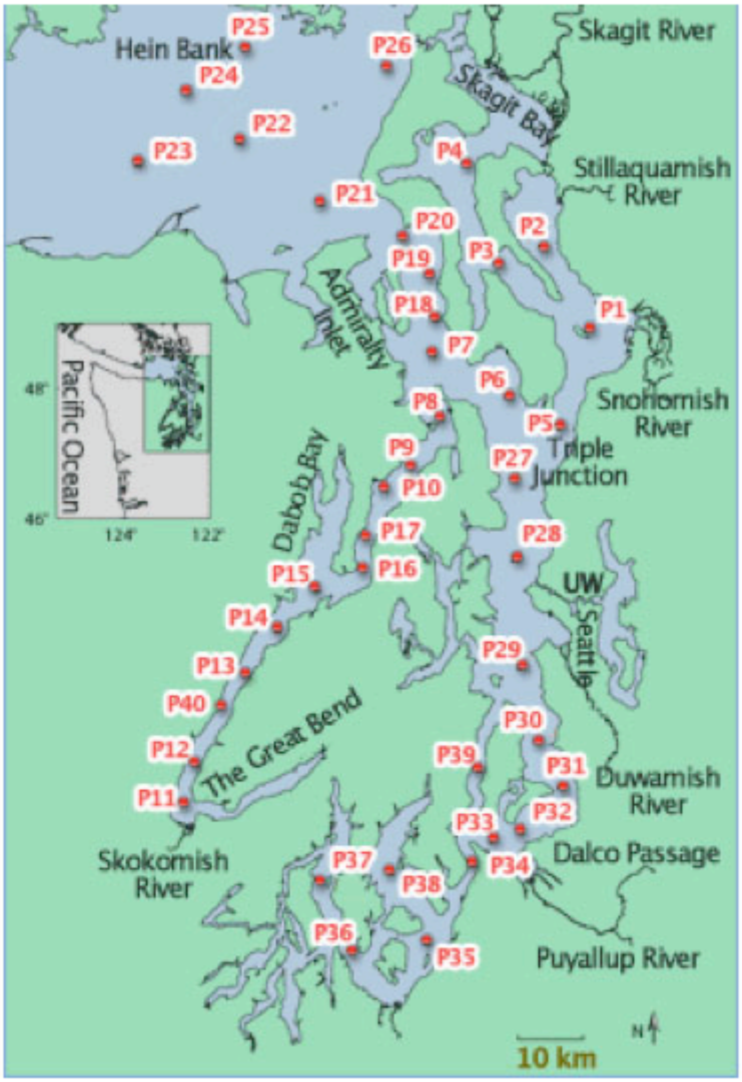


Figure 5. Location of PRISM sampling sites. Used with permission from University of Washington, publisher of web pages for Hood Canal Dissolved Oxygen Program.

Status and Trends

Several groups synthesize monitoring information to evaluate the status of eutrophic conditions throughout Puget Sound and in specific regions that are particularly vulnerable to eutrophication. King County uses a modified version of the Oregon Water Quality Index (Cude 2001) to combine information on dissolved oxygen, DIN and strength of vertical stratification to derive a single number that can be used to assess high to moderate eutrophication risk. In central Puget Sound, index values have been low since 2004 (the first year the index was calculated), except for 2007 when about 20% of the sampling sites showed moderate or high risk. We are unaware of any review process that evaluates the effectiveness of this modified index for predicting the onset of eutrophic conditions. The Department of Ecology published regular updates of their monitoring program up to 2002 (Newton et al. 2002) but no longer continues that reporting format. The Department of Ecology internet portal provides direct access to monitoring data and the results of a ranking algorithm by area for multiple water and sediment quality metrics (Washington State Department of Ecology 2006a).The most recent assessment year is 2008 and the 2010 assessment is scheduled to be complete by September 2010. Briefly, this index scores sample sites on a scale from 1 to 5. Scores of 1 to 3 indicate no water quality impairment, while scores of 4 and 5 indicated impairment. A score of 5 triggers action regarding Total Maximum Daily Loads. No synthetic analysis of the spatio-temporal extent of regions scoring 5 on this scale has been conducted, although the previous iteration of the Puget Sound Science Update reported DO levels at DOE monitoring stations that had very low (< 3 mg /l), low (2 mg /l = 5 mg / l) and high (>5 mg /l) DO levels. In a review of estuarine conditions nationwide, Bricker et al. (2007) reported moderate to high levels of eutrophication in several regions of Puget Sound and high risk for worsening conditions in Hood Canal and South Puget Sound (Table 1). These rankings are based on surveys rather than an explicit and consistent data analysis effort. Albertson et al. (2002) demonstrated eutrophication symptoms in several regions throughout south Puget Sound (Figure 6). Eutrophication in southern Hood Canal has been well documented (Newton 2007) (see Dissolved Oxygen).

Table 1. Summary of current status, future outlook and status of influencing factors by location, From Bricker et al. 2007. Status levels and risk are assigned based on surveys of local experts, not on quantitatively defined categories.

	Influencing Factors	Eutrophic Conditions	Future Outlook (risk of worsening conditions)
Central Puget Sound	Unknown	Moderate	Unknown
South Puget Sound	Unknown	Moderate	High
Skagit Bay / Whidbey Basin	Unknown	Moderate	Unknown
Hood Canal	High	High	High
Sequim / Discovery Bay	Unknown	Moderate / High	Unknown
Port Orchard Sound	Unknown	Moderate	Unknown

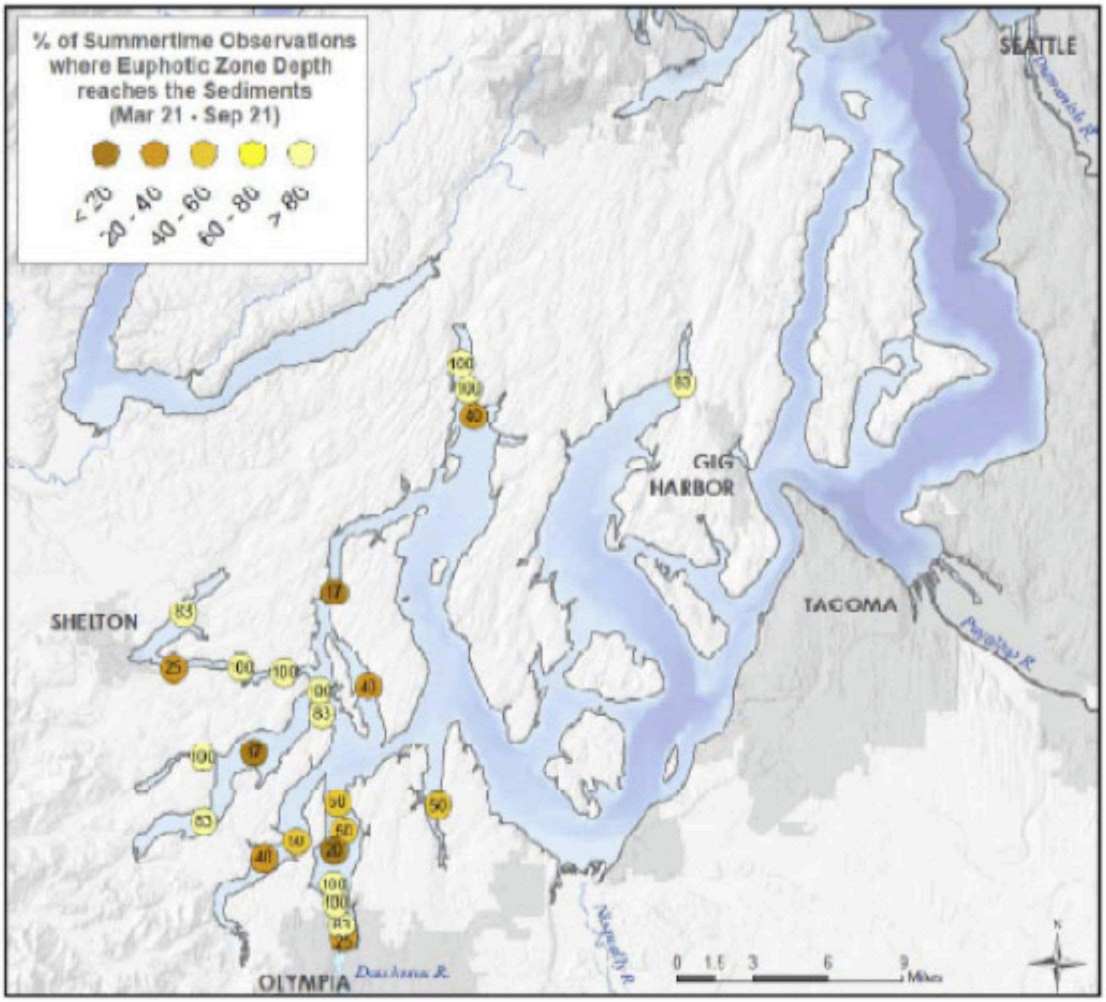


Figure 6. Summertime water clarity in South Puget Sound, 2006 – 2007. Dark points indicate sites with reduced frequencies of high water clarity. Reprinted from Albertson et al. (2002) with permission from Washington Department of Ecology.

Uncertainties

Ongoing research is working to develop detailed biophysical models of Puget Sound that will be useful for gauging the contributions of human activities to changes in trophic status of Puget Sound (Albertson et al. 2007b) and for identifying the most effective policy interventions to prevent worsening conditions. Our present understanding of the threats to Puget Sound is sufficient for identifying areas at risk of cultural eutrophication on the basis of stratification intensity and surface water residence time. We are aware that the Washington State Department of Ecology is presently developing a novel water quality index that may be effective in gauging the current water quality status throughout Puget Sound. Mapping this and other indices against the indicators used in NOAA's national assessment may permit comparisons across ecosystems to better gauge the status of Puget Sound. Future eutrophication status may be affected by climate change through its effects on coastal upwelling intensity, ambient air temperature and timing of freshwater flows.

Literature Cited

Albertson, S., J. Bos, K. Erickson, C. Maloy, G. Pelletier, and M. Roberts. 2007a. South Puget Sound Water Quality Study Phase 2: Dissolved Oxygen Quality Assurance Project Plan. Washington State Department of Ecology.

Albertson, S. L., J. Bos, G. Pelletier, and M. Roberts. 2007b. Estuarine flow in the south basin of Puget Sound and its effects on near-bottom dissolved oxygen. WA Department of Ecology, Olympia, WA.

Albertson, S. L., K. Erickson, J. A. Newton, G. Pelletier, R. A. Reynolds, and M. Roberts. 2002. South Puget Sound Water Quality Study Phase 1. WA Department of Ecology, Olympia, WA.

Bricker, S., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner. 2007. Effects of nutrient enrichment in the nation's estuaries : a decade of change. National Centers for Coastal Ocean Science, Silver Spring MD.

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications 8:559-568.

Cude, C. G. 2001. Oregon water qualit index: a tool for evaluating water quality management effectiveness. Journal fo the American Water Resources Association 37:125-137.

Mackas, D. L. and P. J. Harrison. 1997. Nitrogenous nutrient sources and sinks in the Juan de Fuca Strait / Strait of Georgia / Puget Sound Estuarine system: Assessing the potential for eutrophication. Estuarine, Coastal and Shelf Science 44:1-44.

Newton, J. 2007. Hood Canal: the complex factors causing low dissolved oxygen events require ongoing research, monitoring and modeling. Pages 95-96 in S. Bricker, B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner, editors. Effects of nutrient enrichment in the nation's estuaries : a decade of change. National Centers for Coastal Ocean Science, Silver Spring, MD.

Newton, J. and K. Van Voorhis. 2002. Seasonal patterns and controlling factors of primary production in Puget Sound's central basin and Possesion Sound. WA Department of Ecology, Olympia, WA.

Newton, J. A., S. L. Albertson, K. Van Voorhis, C. Maloy, and E. Siegel. 2002. Washington state marine water quality, 1998 through 2000. Washington State Department of Ecology, Olympia, WA.

Stark, K., S. Mickelson, and S. Keever. 2008. 2005 - 2007 Water Quality Status Report for Marine Waters. King County Department of Natural Resources and Parks, Seattle, WA.

U.S. Environmental Protection Agency. 2006. Puget Sound Georgia Basin ecosystem indicators report: technical background document. U.S. Environmental Protection Agency Region 10.

Washington State Department of Ecology. 2006a. Water Quality Program Policy: Assessment of water quality for the clean water act sections 303(d) and 305(b0 Integrated report. Washington State Department of Ecology, Olympia, WA.

Washington State Department of Ecology. 2006b. Water Quality Program Policy: Ensuring credible data for water quality management. Washington State Department of Ecology, Olympia, WA.

Marine Fecal Bacteria

Background

Fecal bacteria are found in the feces of humans and other homeothermic animals. They are monitored in recreational waters because they are good indicators of harmful pathogens that are more difficult to measure. The two types of fecal bacteria monitored in Puget Sound are fecal coliforms (including E. coli), which are gram-negative rod-shaped bacteria, and enterococci, which are gram-positive spherical bacteria. While fecal coliforms are more commonly monitored, enterococci are also measured because they have higher survival in salt water than coliforms and because they are thought to be more tightly associated with pathogens harmful to humans (Wymer et al. 2005). In Puget Sound, fecal pollution comes from both point-source origins such as combined sewer overflows and direct marine effluent discharge as well as non point-source origins such as surface water runoff, both of which increase with rainfall and river and stream discharge. In addition to serving as an indicator of pathogens, fecal bacterial pollution can also be an indicator of nutrient loading because sewage often contains high levels of nitrogen and phosphorous (Taslakian and Hardy 1976, Costanzo et al. 2001). Both point source (failing septic systems) and non-point sources (landscape features) contribute to fecal bacterial levels in Puget Sound. Additionally, shoreline and basin hydrology can affect the degree of retention of fecal coliform pollution such that bacteria may dissipate more slowly in enclosed bays with diminished water turnover. There currently are approximately 60 permitted wastewater treatment discharge locations in Puget Sound (Stark et al. 2009) (Figure 1) as well as numerous other storm drain and outfall locations.

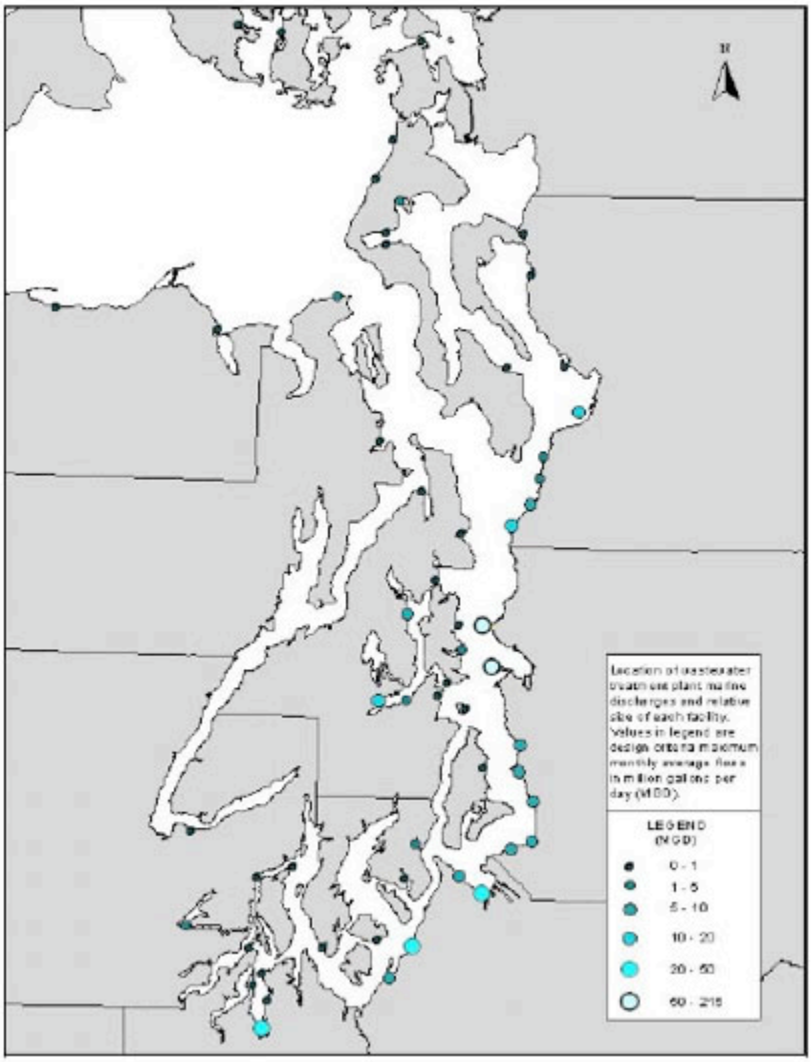


Figure 1. Puget Sound wastewater treatment plant marine discharge locations (reprinted form Stark et al. 2009 with permission from King County Department of Natural Resources and Parks).

In Puget Sound, monitoring of fecal bacteria is conducted by the Washington Department of Health, the Washington Department of Ecology and King County as part of the Puget Sound Ambient Monitoring Project (PSAMP) as well as other local municipalities. The Department of Ecology conducts monthly offshore surveys and assesses both fecal coliforms and enterococci at approximately 40 permanent stations along with a suite of locations that rotate each year (Janzen 1992, Newton et al. 2002)(Figure 2). The Department of Health (DOH) monitors fecal coliforms at 97 commercial shellfish growing areas in Puget Sound (Figure 3). The King County Department of Natural Resources and Parks monitors a combination of inshore and offshore targeted point-source (waste-water discharge) and ambient stations throughout central Puget Sound. The EPA-funded and jointly run (Departments of Health and Ecology) Beach

Environmental Assessment, Communication and Health (BEACH) program monitors and reports on enterococci levels at marine swimming beaches throughout Puget Sound.

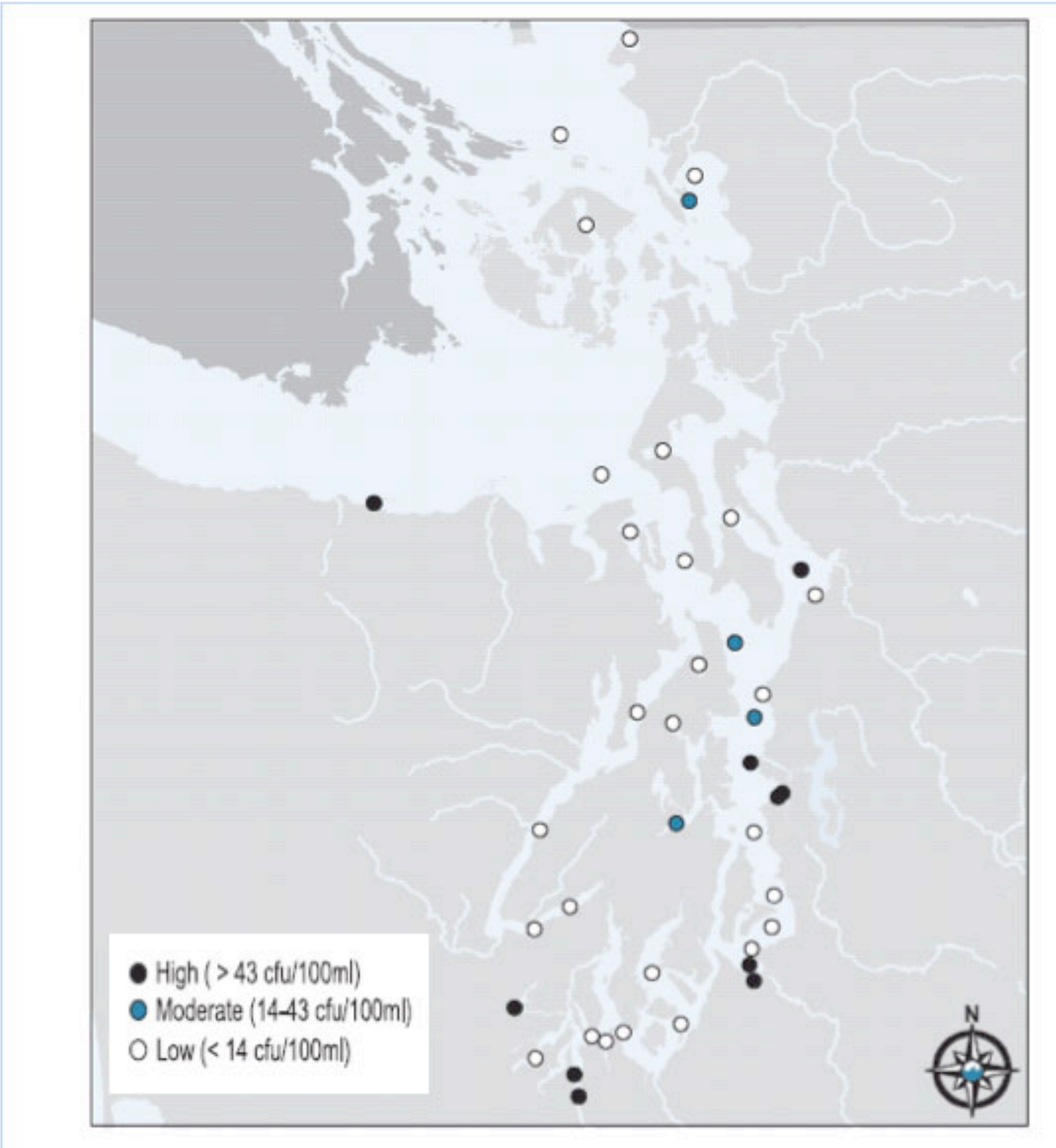
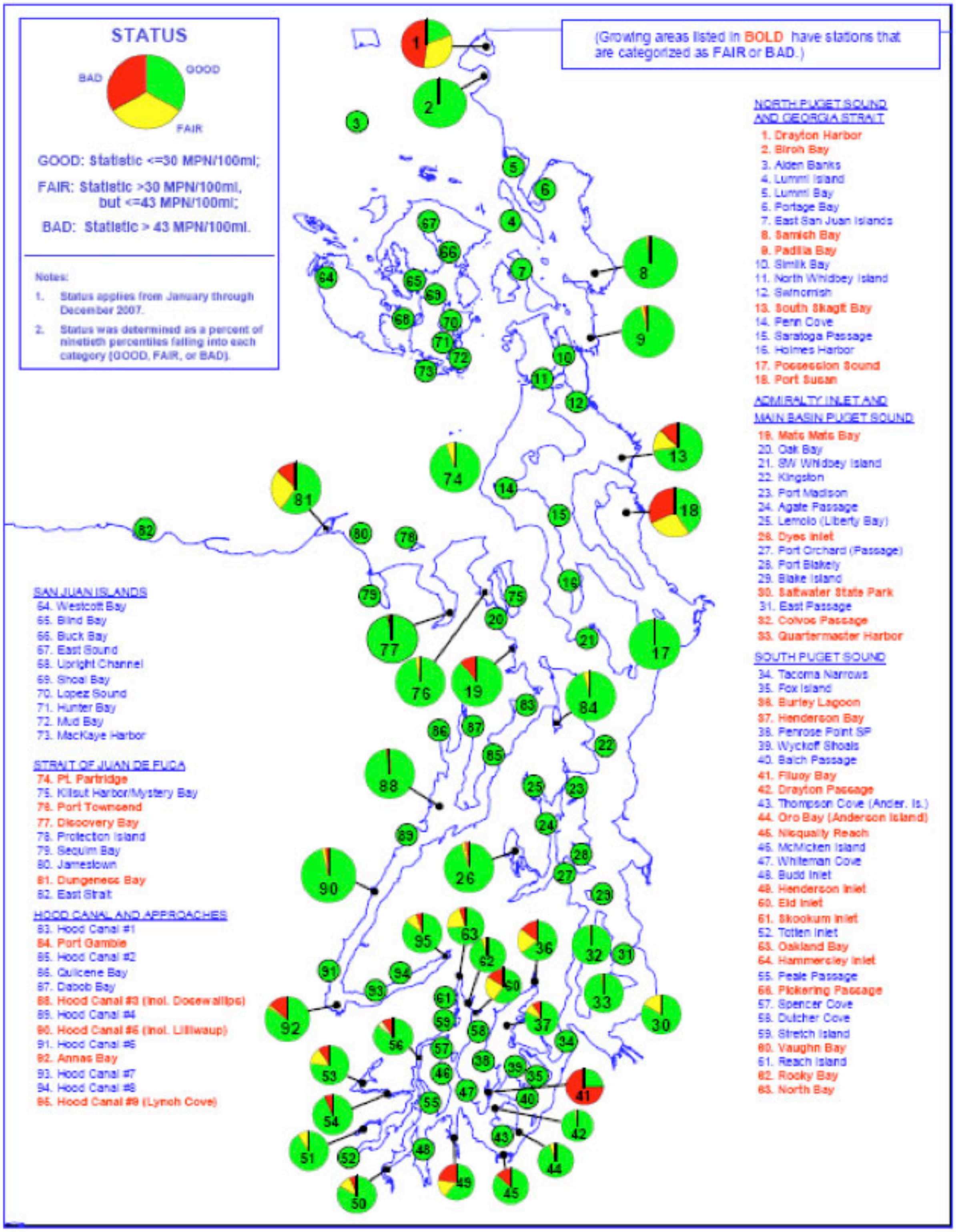


Figure 2. Department of Ecology Marine Waters monitoring stations and maximum fecal coliform bacteria levels (High, Moderate and Low detected Colony Forming Units) from 2001 – 2005 (reprinted from PSP 2007; methodology from Janzen 1992, Newton et al. 2002).



STATUS
BAD
GOOD
FAIR
GOOD: Statistic <=30 MPN/100ml;
FAIR: Statistic >30 MPN/100ml, but <=43 MPN/100ml;
BAD: Statistic > 43 MPN/100ml.
Notes:
1. Status applies from January through December 2007.
2. Status was determined as a percent of ninetieth percentiles falling into each category (GOOD, FAIR, or BAD).
(Growing areas listed in BOLD have stations that are categorized as FAIR or BAD.)
NORTH PUGET SOUND AND GEORGIA STRAIT
1. Drayton Harbor
2. Birch Bay
3. Alden Banks
4. Lummi Island
5. Lummi Bay
6. Portage Bay
7. East San Juan Islands
8. Samish Bay
9. Padilla Bay
10. Similk Bay
11. North Whidbey Island
12. Swinomish
13. South Skagit Bay
14. Penn Cove
15. Saratoga Passage
16. Holmes Harbor
17. Possession Sound
18. Port Susan
ADMIRALTY INLET AND MAIN BASIN PUGET SOUND
19. Mats Mats Bay
20. Oak Bay
21. SW Whidbey Island
22. Kingston
23. Port Madison
24. Agate Passage
25. Lemolo (Liberty Bay)
26. Dyes Inlet
27. Port Orchard (Passage)
28. Port Blakely
29. Blake Island
30. Saltwater State Park
31. East Passage
32. Colvos Passage
33. Quartermaster Harbor
SOUTH PUGET SOUND
34. Tacoma Narrows
35. Fox Island
36. Burley Lagoon
37. Henderson Bay
38. Penrose Point SP
39. Wyckoff Shoals
40. Balch Passage
41. Filucy Bay
42. Drayton Passage
43. Thompson Cove (Ander. Is.)
44. Oro Bay (Anderson Island)
45. Nisqually Reach
46. McMicken Island
47. Whiteman Cove
48. Budd Inlet
49. Henderson Inlet
50. Eld Inlet
51. Skookum Inlet
52. Totten Inlet
53. Oakland Bay
54. Hammersley Inlet
55. Peale Passage
56. Pickering Passage
57. Spencer Cove
58. Dutcher Cove
59. Stretch Island
60. Vaughn Bay
61. Reach Island
62. Rocky Bay
63. North Bay
SAN JUAN ISLANDS
64. Westcott Bay
65. Blind Bay
66. Buck Bay
67. East Sound
68. Upright Channel
69. Shoal Bay
70. Lopez Sound
71. Hunter Bay
72. Mud Bay
73. MacKaye Harbor
STRAIT OF JUAN DE FUCA
74. Pt. Partridge
75. Kilsut Harbor/Mystery Bay
76. Port Townsend
77. Discovery Bay
78. Protection Island
79. Sequim Bay
80. Jamestown
81. Dungeness Bay
82. East Strait
HOOD CANAL AND APPROACHES
83. Hood Canal #1
84. Port Gamble
85. Hood Canal #2
86. Quilcene Bay
87. Dabob Bay
88. Hood Canal #3 (incl. Dosewallips)
89. Hood Canal #4
90. Hood Canal #5 (incl. Lilliwaup)
91. Hood Canal #6
92. Annas Bay
93. Hood Canal #7
94. Hood Canal #8
95. Hood Canal #9 (Lynch Cove)

Figure 3. Commercial shellfish growing areas monitored by the Department of Health in 2007 with fecal pollution levels measured in Most Probable Number (MPN)/100m. Pie charts show the proportion of samples at each location with Good (≤ 30 MPN/100mL), Fair (>3 and ≤ 43 MPN/100mL) and Bad(> 43 MPN/100mL) fecal pollution levels (reprinted from Determan 2009; courtesy of Washington State Department of Health Shellfish Program).

Monitoring by all agencies is conducted with the intent of determining whether bacterial counts meet or exceed established critical levels. For fecal coliforms, the State of Washington (WAC 173-201, 1991) mandates that in class A and AA marine waters, bacterial counts should not exceed a geometric mean of 14 organisms/100mL with no more than 10 % exceeding 43 organisms/100mL (Newton et al. 2002). Similar standards for coliforms are mandated by the National Shellfish Sanitation Program (NSSP) for shellfish growing areas such that the geometric mean of an area cannot exceed 14 organisms/100mL or that the estimated 90th percentile cannot exceed 43 organisms for cases where only non-point sources are present. For enterococci, the minimum advisory standard recommended by the EPA for recreational beaches is 35 colonies/100mL (Schneider 2002, Wymer et al. 2005). Fecal coliform levels are also a component of Federal Clean Water Act standards. Two agencies, the Department of Health (Determan 2009) and King County (Stark et al. 2009), have developed indices to rank sites according to the frequency and intensity of increases above Washington State standards in observed fecal coliform levels.

Status and Trends

The most recently reported assessment of fecal coliforms by the Department of Ecology monitoring program revealed that the highest levels of coliforms were observed in Budd Inlet, Commencement Bay, Oakland Bay, Port Angeles Harbor, Possession Sound and Elliot Bay from 2001 – 2005 (Janzen 1992, methodology from Newton et al. 2002, reported in PSP 2007)(Figure 2). Of the 97 shellfish growing areas tested by the Department of Health in 2007, the highest fecal pollution levels were found in Filucy Bay, Drayton Harbor, Burley Lagoon and Port Susan (Determan 2009)(Figures 3, 4). Using a calculated Fecal Pollution Index, which integrates the frequency and intensity of events of elevated fecal coliform levels and ranges from 1 to 3, they found that the sound-wide FPI was 1.16 (Determan 2009). A trend analysis showed that the sound-wide FPI had not changed significantly from 1998 – 2007 (Determan 2009)(Figure 5). The Frequency of Exceedence (FOE) index of fecal coliform bacteria utilized by the King County shellfish area monitoring program identified Alki Point, Shilshole Bay, Fauntleroy Cove, Magnolia and Inner Elliott Bay as the locations with the highest FOE in 2004 (reported in PSP 2007, methodology from Stark et al. 2009)(Figure 1). The most recent enterococci levels reported by the BEACH program showed that of the 70 beaches monitored in 2004 and 2005, the highest number of exceedances were in locations that were largely on septic systems such as Birch Bay County Park and Bayview State Park, in enclosed bays such as Freeland Park as well as beaches in Sinclair and Dyes Inlets and Twanoh State Park (methodology from Schneider 2004, reported in PSP 2007)(Figure 7).

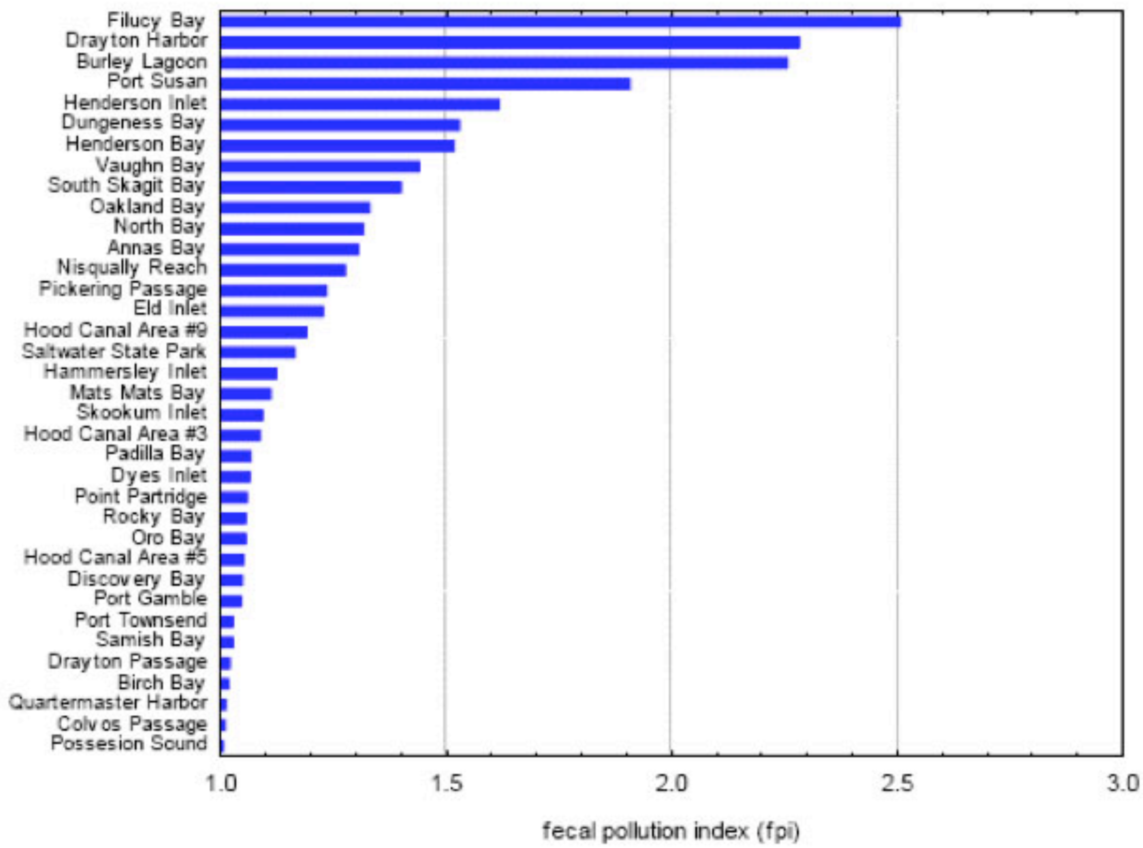


Figure 4. Department of Health rankings of 36 commercial shellfish growing areas in Puget Sound according to the fecal pollution index in 2007 (reprinted from Determan 2009; courtesy of Washington State Department of Health Shellfish Program).

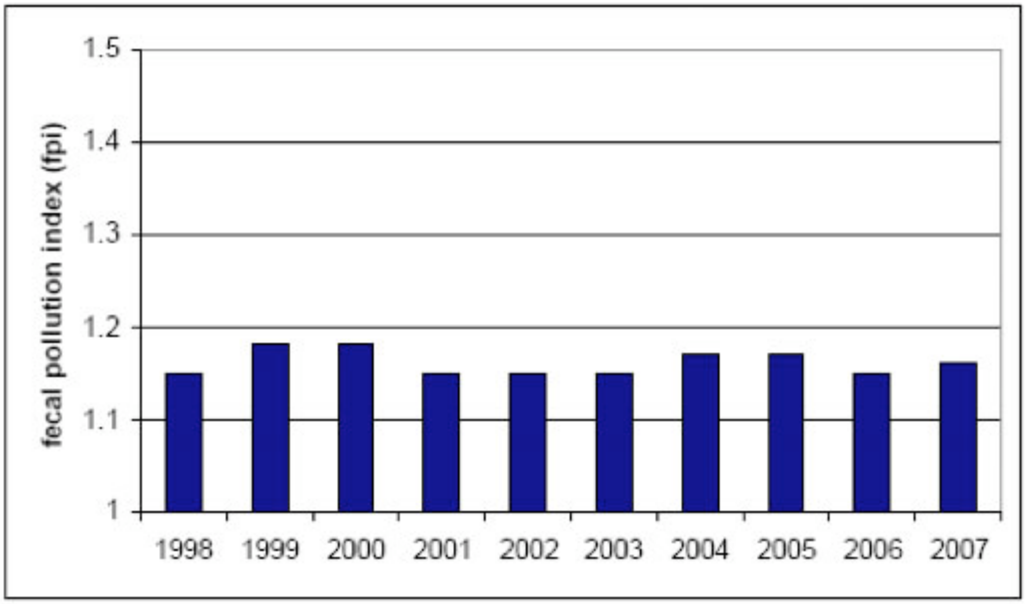


Figure 5. Fecal pollution index at commercial growing areas monitored by the Department of Health in Puget Sound from 1998 – 2007 (reprinted from Determan 2009; courtesy of Washington State Department of Health Shellfish Program).

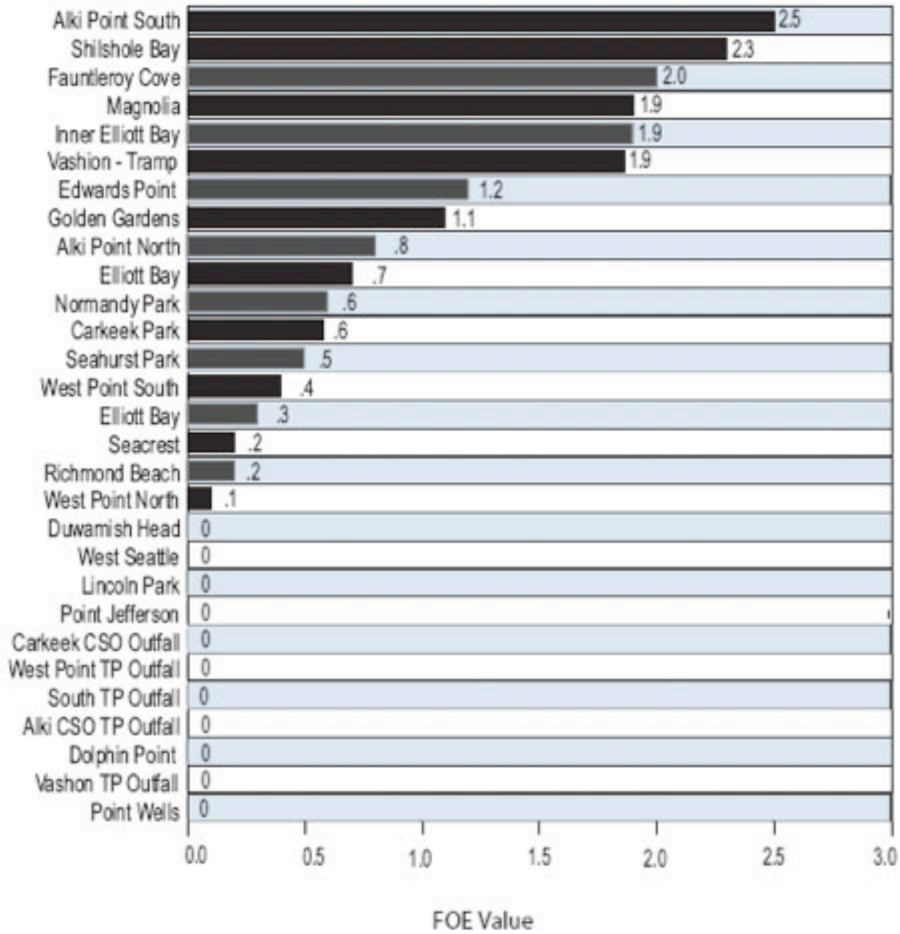


Figure 6. Frequency Of Exceedence (FOE) index of fecal coliform bacteria from offshore and beach stations monitored by King County Department of Natural Resources and Parks in 2004 (reprinted from PSP 2007; methodology from Stark et al. 2009).

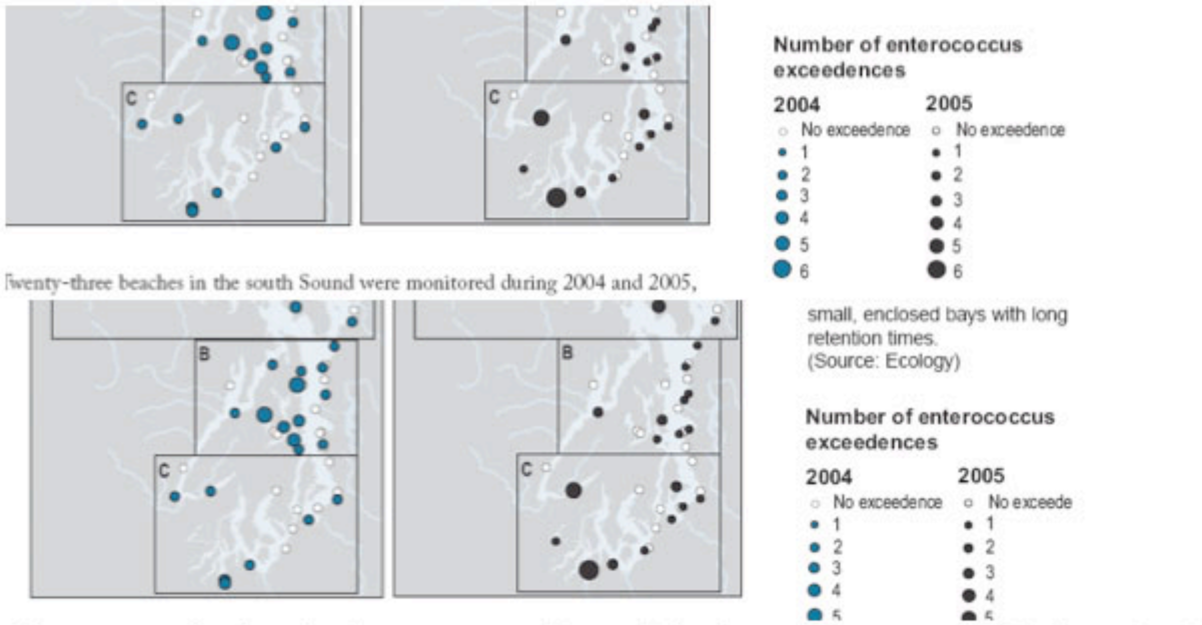


Figure 7. Monitoring sites for enterococci bacteria by the BEACH program (jointly run by the Department of Ecology and the Department of Health) and the number of times enterococci levels location exceeded program-defined guidelines (reprinted from PSP 2007; methodology from Schneider 2002, 2004).

Uncertainties

While fecal coliform levels in Puget Sound are well documented, disparate data sources make understanding broad spatial and temporal trends challenging, thereby obscuring potentially important patterns. Local hydrology, water temperature and salinity may all affect the persistence of fecal coliforms in Puget Sound yet this has not been investigated. Finally, the relative contribution of pet waste to overall fecal coliforms levels in Puget Sound has not been examined yet disease transfer from domestic pets to mammalian wildlife by this mechanism has been shown in other systems (Miller et al. 2002).

Summary

Considerable monitoring effort contributes to the assessment of fecal bacteria in Puget Sound. No single area or basin of Puget Sound was identified as consistently having the highest fecal coliform levels. A single analysis evaluating spatial and temporal trends based on all available data sources for fecal bacteria in Puget Sound has not been conducted.

Literature Cited

Costanzo, S. D., M. J. O'Donohue, W. C. Dennison, N. R. Loneragan, and M. Thomas. 2001. A new approach for detecting and mapping sewage impacts. Marine Pollution Bulletin 42:149-156.

Determan, T. 2009. Atlas of Fecal Coliform Pollution in Commercial Shellfish Areas of Puget Sound: Year 2007. Washington State Department of Health, Olympia, WA.

Janzen, C. 1992. Marine Water Column Ambient Monitoring Plan. Washington State Department of Ecology, Olympia, WA.

Miller, M. A., I. A. Gardner, C. Kreuder, D. M. Paradies, K. R. Worcester, D. A. Jessup, E. Dodd, M. D. Harris, J. A. Ames, A. E. Packham, and P. A. Conrad. 2002. Coastal freshwater runoff is a risk factor for *Toxoplasma gondii* infection of southern sea otters (*Enhydra lutris nereis*). International Journal for Parasitology 32:997-1006.

Newton, J. A., S. L. Albertson, k. V. Voorhis, C. Maloy, and E.Siegel. 2002. Washington State Marine Water Quality, 1998 through 2000. Washington State Department of Ecology, Olympia, WA.

PSP. 2007. Puget Sound Update: Ninth Report of the Puget Sound Assessment and Monitoring Program. Page 260, Seattle, WA.

Schneider, L. 2002. Draft BEACH Program Guidance. Washington State Department of Ecology, Olympia, WA.

Schneider, L. 2004. Quality Assurance Project Plan: BEACH Program. Washington State Department of Ecology, Olympia, WA.

Stark, K., S. Mickelson, and S. Keever. 2009. Water Quality Status Report for Marine Waters, 2005 - 2007. King County Department of Natural Resources and Parks, Seattle, WA.

Taslakian, M. J., and J. T. Hardy. 1976. Sewage nutrient enrichment and phytoplankton ecology along the central coast of Lebanon. Marine Biology 38:315-325.

Wymer, L. J., K. P. Brenner, J. W. Martinson, W. R. Stutts, S. A. Schaub, and A. P. Dufour. 2005. The EMPACT Beaches Project: Results from a study on microbiological monitoring in recreational waters. U.S. Environmental Protection Agency, Cincinnati, OH.

Water Quantity

Here we provide a limited synthesis of stream gauge data to examine trends in freshwater flows with respect to annual and daily flows, timing of flow, low flows and flows relative to instream flow guidelines. This is intended to supplement a review of published information, but we caution that a full analysis of these data and appropriate vetting of methods and interpretations is needed to fully assess the status of freshwater flows. It is our intention that this data compilation and analysis be used to identify data limitations and other key uncertainties with respect to the Puget Sound Partnership Water Quantity Priorities.

Data sources

There are approximately 90 gauging stations overseen by the United States Geological Survey (USGS) in the Puget Sound basin that are located on unregulated reaches of rivers and streams that may be suitable for the analysis of streamflow status and trends (United States Geological Survey 2010b). A complete analysis of all available data was not performed for this report. Instead, data from at least one unregulated gauging station within each Water Resource Inventory Area (WRIA) were included whenever possible. This selection was based on the intent to capture broad regional coverage.

We included all data from available gauging stations on unregulated reaches in the Skagit River basin in order to determine whether there were basin-wide correlations in the hydrologic indicators. Previous reports have combined streamflow data from several rivers to evaluate regional trends (Puget Sound Partnership 2009). A strong correlation between stream and rivers within the same basin could suggest that this is a valid approach. We review evidence for correlation here using simple descriptive statistics, but emphasize that a more rigorous analysis is warranted.

1. Flow Timing

Background

Puget Sound river hydrology could be affected by climate change. Precipitation in the region occurs predominately in the winter months. The accumulation of snow in the mountains is a primary storage mechanism, particularly for the snowmelt-dominated and transitional river systems. It has been estimated that more than 70% of total stream discharge in the Western United States is from melting snowpack (1996). An estimated 27% of summer streamflow of the Nooksack River originates from high-elevation snowshed and glacier-derived meltwater (Bach 2002). Climate change assessments predict increased winter and spring temperatures, resulting in decreased snowpack storage in the mountains, increased winter runoff as more precipitation falls as rain, and lower summer flows (Hamlet and Lettenmaier 1999, Lettenmaier et al. 1999, Mote et al. 1999, Leung et al. 2004, Barnett et al. 2008). Climate change may force rivers with snowmelt-dominated and transitional hydrological flow patterns towards rainfall-dominated hydrology (Mote et al. 1999). These changes are measurable through flow timing metrics, including the timing of the center of mass of annual flow (CT).

Prediction of the regional impacts of climate change on river and stream hydrology can be confounded by typical variation in rainfall patterns, high geographic variability, and land use changes. At least two large-scale systems affect annual climate variations in the Pacific Northwest (Mote 2003). These are the El Niño/Southern Oscillation, with a period of 2 to 7 years, and the Pacific Decadal Oscillation (PDO), with an estimated period of 20 to 30 years. Warm and cool phases of the El Niño/Southern Oscillation and/or Pacific Decadal Oscillation can result in variations on the order of 1°C for temperature, and 20% for precipitation (Mote et al. 2003). Hamlet et al. (2005) utilized a Variable Infiltration Capacity model to discern long-term trends in spring snowpack and snowmelt timing, decadal temperature and precipitation variability. They found that the date on which 90% snowmelt occurred correlated strongly with winter temperatures in the Pacific Northwest, and that there was a long-term warming trend that was not associated with decadal oscillations. In a subsequent study, Hamlet et al. (2007) specifically investigated the relationship between temperature, precipitation, and runoff timing in the western United States and found that in warmer areas, including the Pacific Northwest, fractional streamflow tended to occur earlier in the year (Hamlet et al. 2007). Mote et al. (2008) concluded that the primary factor in decreasing snowpack in the Washington Cascades was rising temperatures, consistent with the global warming. The long-term snowpack trends were unrelated to the variability brought about by Pacific oscillations.

Stewart et al. (2004) investigated historical (1948-2000) and projected future streamflow timing in snowmelt dominated rivers and streams in the Western United States. They found significant trends towards earlier runoff in many rivers and streams in the Pacific Northwest. Utilizing a ‘business-as-usual’ emissions scenario with a Parallel Climate Model, they predicted a continuation of this trend, largely due to increased winter and spring temperatures, but not changes in precipitation. In a companion study they further analyzed the trends in streamflow timing with variations of the PDO (Stewart et al. 2005). While streamflow timing was partially controlled by the PDO, there remained a significant part of the variation in timing that was explained by a longer-term warming trend in spring temperatures. This suggests that earlier seasonal flows may be associated with warming.

In addition to accelerated spring snowmelt, the shift toward earlier runoff timing can be attributed to a larger fraction of winter precipitation occurring as rain instead of snow. Knowles et al.(2006) evaluated data from the western United States and found a decreasing fraction of winter precipitation falling as snow. The largest decreases occurred in warmer winter areas, such as the Pacific Northwest, where relatively small increases in temperature would result in the transition from snowfall to rainfall, resulting in less snowpack and earlier runoff timing (Knowles et al. 2006).

Using a multivariate analysis, Barnett et al. (2008) evaluated simultaneous changes in average winter temperature, snow pack, and runoff timing in the Western United States (including the Washington Cascades) for the period from 1950 – 1999. They found significant increasing trends in winter temperature, and decreasing trends in snow pack and runoff timing (indicating earlier snowmelt) and that this was mostly like driven by anthropogenic forcing (Barnett et al. 2008).

Recently, the Climate Impacts Group at the University of Washington performed The Washington Climate Change Impact Assessment. The assessment included analyses of hydrology and water resource management in which they utilized results from 20 global climate models and two emissions scenarios from the IPCC Special Report on Emissions Scenarios (A1B and B1) to evaluate projected changes in spring snowpack and runoff (Elsner et al. 2009). For the rivers in the Puget Sound basin they found a dramatic decrease in spring snowpack, with almost no April 1 snowpack by 2080. During that period, river hydrographs progressively changed from transition or snow-rain dominated to rain dominated patterns. There was little predicted change in annual precipitation.

The observed and predicted changes in river flow regime described above can affect water resource management in the Pacific Northwest where systems have been designed based on historical flow patterns (Lettenmaier et al. 1999, Milly et al. 2008). Wiley and Palmer(2008) utilized a three-stage modeling approach to evaluate the potential impacts of climate change on the Seattle water supply system. They found a decreasing annual system yield (the amount of water that can be reliably supplied by a system) largely due to earlier runoff and decreasing water storage in the mountain snowpack. Vano et al. (2009) expanded this analysis to include the Everett and Tacoma water systems. They found that altered flow regimes likely will reduce the reliability of all three systems, particularly in the face of increasing demand, and could affect ancillary operations such as flood control, power generation, and the augmentation of environmental flows.

Several measures of flow timing exist. One measure of river flow timing is centroid timing (CT), calculated by Stewart et al. (2005) and Elsner et al. (2009):

where: qi=daily flow and ti=number of days past the beginning of the water year.

The centroid of flow measure is relatively insensitive to false interannual variations, is easy to calculate, and allows for easy comparisons of basins (Stewart et al. 2004).

There are approximately 90 gauging stations overseen by the United States Geological Survey (USGS) in the Puget Sound basin that are located on unregulated reaches of rivers and streams, which may be suitable for the analysis of streamflow status and trends (USGS Water Center); a list is provided in Chapter 1 of this report. A complete analysis of all of the available data was not performed for the purposes of this report. However, data from at least one unregulated gauging station within each Water Resource Inventory Area (WRIA) was included where possible in order to coarsely approximate a regional scale.

Data from all available gauging stations on unregulated reaches in the Skagit River basin were included in this analysis in order to evaluate whether there existed any basin-wide correlations in the hydrologic indicators. Previous reports have combined streamflow data from several rivers to evaluate regional trends (PSP 2009). A strong correlation between stream and rivers within the same basin would indicate that this is a valid approach.

Status

Centroid timing values were calculated using gauge data from 14 different locations on unregulated rivers within the Puget Sound basin, in order to evaluate the status and trends of streamflow timing within the region. The results are shown in Table 1. The Pearson's Correlation Coefficients for the annual CT are shown in Table 2.

Table 1. Calculated centroid of flow timing (CT) and trends in CT for unregulated rivers and streams in the Puget Sound

			Centroid of Annual Flow		
River	**Data Years**		**Average Date**	**Annual Change (days/year)**	**p (change ≠ 0)**
WRIA 1 – Nooksack					
Nooksack USGS 12213100	1966-2009		3/18	-0.2±0.14	0.13
WRIA 3/4 – Upper-Lower Skagit and Samish					
Lower Sauk USGS 12189500	1936-2009		4/4	-0.20±0.08	0.01
Upper Sauk USGS 12186000	1929-2009		4/2	-0.17±0.08	0.03
Thunder USGS 12175500	1931-2009		5/16	-0.07±0.06	0.23
Newhalem USGS 12178100	1962-2009		4/10	-0.40±0.17	0.02
Samish USGS 12201500	1945-1970 1996-2009		2/13	-0.01±0.08	0.85
WRIA 5 - Stillaguamish					
Stillaguamish USGS 12167000	1929-2009		2/26	-0.13±0.06	0.05
WRIA 7 – Snohomish					
Skykomish USGS 12134500	1929-2009		3/22	-0.17±0.08	0.04
WRIA 8 – Cedar/Sammamish					
Cedar USGS 12114500	1947-2009		3/16	-0.15±0.13	0.23
WRIA 10 – Puyallup/White					
Puyallup USGS 12092000	1957-2009		4/6	0.01±0.13	0.94
WRIA 11 - Nisqually					
Nisqually USGS 12082500	1942-2009		3/30	-0.11±0.08	0.22
WRIA 13 - Deschutes					
Lower Deschutes USGS 12080010	1946-1963 1990-2009		2/22	0.00±0.07	0.97
Upper Deschutes USGS 12079000	1950-2009		2/14	0.02±0.09	0.80
WRIA 16 – Skokomish/Dosewalips					
Duckabush USGS 12054000	1939-2009		3/14	-0.11±0.09	0.20
Notes:	1. Center of Flow is calculated by: $CT = \frac{\sum (q_i t_i)}{\sum q_i}$				

Table 2. Pearson's Correlation Coefficient for annual CT for rivers within WRIA 3/4.

	Lower Sauk	Upper Sauk	Thunder	Cascade	Newhalem	Samish
Lower Sauk		0.98	0.85	0.97	0.94	0.59
Upper Sauk			0.85	0.96	0.95	0.52
Thunder				0.88	0.85	0.54
Cascade					0.88	0.59
Newhalem						0.65

Note: All Peasrosn's correlation coefficients are significantly different than 0.

There appears to be a relatively strong correlation for this particular metric in flows within the Skagit River basin (r>0.85). The correlation between the rivers in the Skagit River Basin and the Samish River is less robust (r<0.65).

Trends

Annual CT values were calculated for the water years with complete data sets for 14 gauge stations in the Puget Sound. The trend of CT versus time was determined using simple linear regression. The significance of the trends were determined by evaluating the probability that the slope of the trendline was significantly different than zero. Results are shown in Table 1. The rivers with significant trends (P<0.05; Lower Sauk, Upper Sauk, Newhalem, NF Stillaguamish, and Skykomish) all showed an annual decrease in flow timing indicating that peak flows occur earlier in the year (Table 1). There were no rivers with significant trends indicating later flows. Overall, the centroid of flow timing occurred from 1.5-4 days earlier per decade. Data from two of the three rainfall-dominated river systems (Samish and Deschutes) and the single snowmelt-dominated river (Thunder) indicated no significant change in streamflow timing (P>0.05; Table 1).

Uncertainties

The analysis presented above was derived from data in the public domain. The values and trends for CT were calculated from average daily discharge data from USGS station located in the Puget Sound region (United States Geological Survey 2010b). The datasets include qualification codes indicating whether data are provisional or have been approved (United States Geological Survey 2010a). We avoided using provisional data in this analysis, and we omitted data from gauging stations for which advisory notes warning against unreliable data quality had been posted. Average daily discharge data for each water year (October 1 – September 30) were used to calculate the CT. The existence of trends was determined by evaluating the probability of the slope of the CT versus year, as determined through simple linear regression; trends were those with slope significantly different than zero (P<0.05).

Due to interannual variation, the selection of the beginning and ending years of streamflow data may affect the significance of the trend reported in Table 1 . Konrad et al. (2002) used both parametric and nonparametric tests and found a high likelihood of Type I errors when using 10-year streamflow records to evaluate long-term trends. In this evaluation we used a minimum record length of 37 years; the shortest record that resulted in a significant trend was 47 years.

The significance of the Pearson's correlation coefficient was determined by calculating the probability that the correlation was different than zero based on the value of the correlation and the sample size. A significant correlation does not indicate a strong correlation.

Summary

Of the fourteen data sets analyzed, four showed significant decreasing trends, indicating flow timing earlier in the water year. The rate of timing change was from 1.5-4 days per decade. The other ten data sets showed no significant trends.

There was significant variation in the flow timing data sets. However, there was a strong correlation in CT between rivers within the Skagit River basin (Pearson's r>0.85). The correlation between the CT of the Samish river and the rivers in the Skagit River basin was weaker (Pearson's r<0.65).

The CT could be a useful indicator of hydrologic alteration. It allows the tracking of potential changes due to climate, allows comparison of trends across different river systems, and is of importance to water resources managers. It may be more valuable when combined with other indicators of hydrologic alteration to give a more complete picture of streamflow patterns.

Literature Cited

Bach, A. 2002. Snowshed contributions to the Nooksack river watershed, North Cascades Range, Washington. Geographical Review 92:192-212.

Barnett, T. P., D. W. Pierce, H. G. Hidalgo, C. Bonfils, B. D. Santer, T. Das, G. Bala, A. W. Wood, T. Nozawa, A. A. Mirin, D. R. Cayan, and M. D. Dettinger. 2008. Human-induced changes in the hydrology of the western United States. Science 319:1080-1083.

Cayan, D. R. 1996. Interannual climate variability and snowpack in the western United States. Journal of Climate 9:928-948.

Elsner, M. M., L. Cuo, N. Voisin, J. S. Deems, A. F. Hamlet, J. A. Vano, K. E. Mickelson, S.-Y. Lee, and D. P. Lettenmaier. 2009. Implications of 21st Century Climate Change for the Hydrology of Washington State. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Hamlet, A. F., and D. P. Lettenmaier. 1999. Effects of climate change on hydrology and water resources in the Columbia River basin. Journal of the American Water Resources Association 35:1597-1623.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2005. Effects of temperature and precipitation variability on snowpack trends in the western United States. Journal of Climate 18:4545-4561.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2007. Twentieth-century trends in runoff, evapotranspiration, and soil moisture in the western United States. Journal of Climate 20:1468-1486.

Knowles, N., M. D. Dettinger, and D. R. Cayan. 2006. Trends in snowfall versus rainfall in the Western United States. Journal of Climate 19:4545-4559.

Konrad, C. P., and D. Booth. 2002. Hydrologic Trends Associated with Urban Development for Selected Streams in the Puget Sound Basin, Western Washington. U.S. Geological Survey, Tacoma, WA.

Lettenmaier, D. P., A. W. Wood, R. N. Palmer, E. F. Wood, and E. Z. Stakhiv. 1999. Water resources implications of global warming: A US regional perspective. Climatic Change 43:537-579.

Leung, L. R., Y. Qian, X. D. Bian, W. M. Washington, J. G. Han, and J. O. Roads. 2004. Mid-century ensemble regional climate change scenarios for the western United States. Climatic Change 62:75-113.

Milly, P. C. D., J. Betancourt, M. Falkenmark, R. M. Hirsch, Z. W. Kundzewicz, D. P. Lettenmaier, and R. J. Stouffer. 2008. Climate change - Stationarity is dead: Whither water management? Science 319:573-574.

Mote, P., D. Canning, D. Fluharty, R. Francis, J. Franklin, A. Hamlet, M. Hershman, M. Holmberg, K. G. Ideker, W. Keeton, D. Lettenmaier, R. Leung, N. Mantua, E. Miles, B. Noble, H. Parandvash, D. W. Peterson, A. Snover, and S. Willard. 1999. Impacts of climate variability and change inthe Pacific Northwest. The JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Mote, P., A. Hamlet, and E. Salathe. 2008. Has spring snowpack declined in the Washington Cascades? Hydrology and Earth System Sciences 12:193-206.

Mote, P. W. 2003. Trends in snow water equivalent in the Pacific Northwest and their climatic causes. Geophysical Research Letters 30.

Mote, P. W., E. Parson, A. F. Hamlet, W. S. Keeton, D. Lettenmaier, N. Mantua, E. L. Miles, D. Peterson, D. L. Peterson, R. Slaughter, and A. K. Snover. 2003. Preparing for climatic change: The water, salmon, and forests of the Pacific Northwest. Climatic Change 61:45-88.

PSP. 2009. Ecosystem Status and Trends. Water Quantity., Puget Sound Partnership.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2004. Changes in snowmelt runoff timing in western North America under a 'business as usual' climate change scenario. Climatic Change 62:217-232.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate 18:1136-1155.

United States Geological Survey. 2010a. Provisional data for Washington. http://waterdata.usgs.gov/wa/nwis/help/?provisional

United States Geological Survey. 2010b. USGS Washington Water Science Center. http://wa.water.usgs.gov/realtime/current.html

Vano, J. A., N. Voisin, L. Cuo, A. F. Hamlet, M. M. Elsner, R. N. Palmer, A. Polebitski, and D. P. Lettenmaier. 2009. Climate Change Impacts on Water Management in the Puget Sound

Region, Washington, USA. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Wiley, M. W., and R. N. Palmer. 2008. Estimating the impacts and uncertainty of climate change on a municipal water supply system. Journal of Water Resources Planning and Management-Asce 134:239-246.

Average Annual Flow

Background

Average annual flow rate can be affected by changes in precipitation. Analysis of historical precipitation data suggests that significant trends in historical rainfall patterns associated with climate change in the Pacific Northwest are not detectable (Hamlet et al. 2005, Mote et al. 2005, Hamlet and Lettenmaier 2007, Hamlet et al. 2007). Climate change modeling suggests that there may be only modest increases in annual precipitation by 2080 (Elsner et al. 2009). Annual rainfall has been shown to be correlated with the Pacific Decadal Oscillation and El Nino Southern Oscillation, and variations in rainfall patterns may have increased in recent years (Hamlet and Lettenmaier 2007, Luce and Holden 2009). Increases in the variability of rainfall and streamflow in the Pacific Northwest may put pressure on water supply systems, which were designed based on historical variations (Jain et al. 2005, Hamlet and Lettenmaier 2007). One analysis (Pagano and Garen 2005) suggested that low-flow years were more likely to occur in succession, potentially exacerbating water supply pressures.

Luce and Holden (2009) utilized quartile regression to investigate trends in streamflow in wet (75th percentile), dry (25th percentile), and average (50th percentile) water years in rivers in the Pacific Northwest. They concluded that the dry years were getting dryer in the Pacific Northwest, accounting for much of the increased variability in annual streamflow.

Average annual flow may also be affected by land use changes. Logging in watersheds can reduce evapo-transpiration resulting in increased annual flows (Bosch and Hewlett 1982). Results from modeling studies suggest there is an increase in annual mean streamflow due to land use change in the Puget Sound lowlands (Cuo et al. 2009). The construction of storm drains associated with urbanization may result in lower streamflows (Simmons and Reynolds 1982). Increased diversions and consumptive uses may also result in lower overall streamflows.

Status and Trends

Data from the Cedar River (below Bear Creek, near Cedar Falls) indicated a significant decrease in annual average streamflow from 1946-2009 ($p=0.03$; ca. 0.3% yr-1 decrease; Table 1). No other river systems showed a significant change in annual average streamflow (Table 1). The Pearson's Correlation Coefficients for the average annual flow rate between the river systems in WRIA 3/4 indicate that there is a strong linear correlation between the annual average flow rates of the rivers evaluated ($r>0.83$; Table 2). There was a somewhat weaker correlation ($0.68<r<0.81$) between the Samish River and the rivers of the Skagit River basin,,all of which lie within WRIA 3/4.

Table 1. Average annual flow rate in cubic feet per second (CFS) and annual change in average flow rate as determined by simple linear regression (±standard error). Data from USGS Washington Water Science Center (http://wa.water.usgs.gov/)

			AVERAGE FLOW	
River	**Data Years**		**Average Flow Rate**	**Annual Change**
			(CFS)	(ΔCFS/Year)
WRIA 1 – Nooksack				
Nooksack USGS 12213100	1966-2009		3855	-3.7±9.0
WRIA 3/4 – Upper-Lower Skagit and Samish				
Lower Sauk USGS 12189500	1936-2009		4342	2.0±4.5
Upper Sauk USGS 12186000	1929-2009		1118	0.0±1.1
Thunder USGS 12175500	1931-2009		619	0.2±0.4
Newhalem USGS 12178100	1962-2009		176	0.1±0.3
Samish USGS 12201500	1945-1970 1996-2009		246	0.2±0.4
WRIA 5 - Stillaguamish				
Stillaguamish USGS 12167000	1929-2009		1897	2.8±1.9
WRIA 7 – Snohomish				
Skykomish USGS 12134500	1929-2009		3957	3.5±4.1
WRIA 8 – Cedar/Sammamish				
Cedar USGS 12114500	1947-2009		161	-0.5±0.2
WRIA 10 – Puyallup/White				
Puyallup USGS 12092000	1957-2009		527	-0.4±0.6
WRIA 11 - Nisqually				
Nisqually USGS 12082500	1942-2009		772	-0.0±0.9
WRIA 13 - Deschutes				
Lower Deschutes USGS 12080010	1946-1963 1990-2009		397	0.2±0.9
Upper Deschutes USGS 12079000	1950-2009		258	-0.2±0.7
WRIA 16 – Skokomish/Dosewalips				
Duckabush USGS 12054000	1939-2009		416	0.0±0.5

Table 2. Pearson's Correlation Coefficient of annual average flow rates between river systems in WRIA 3/4. All correlations are significantly different than zero ($P<0.05$).

	Lower Sauk	Upper Sauk	Thunder	Cascade	Newhalem	Samish
Lower Sauk		0.98	0.85	0.97	0.94	0.81
Upper Sauk			0.83	0.97	0.94	0.75
Thunder				0.87	0.86	0.68
Cascade					0.87	0.73
Newhalem						0.73

Uncertainties

This analysis was derived from data within the public domain. Average annual flow data presented were calculated from average daily discharge data from USGS stations located in the Puget Sound region (United States Geological Survey 2010b). The datasets include qualification codes indicating whether data are provisional or have been approved (United States Geological Survey 2010a). We avoided using provisional data in this analysis, and we omitted data from gauging stations for which advisory notes warning against unreliable data quality had been posted.

Average daily discharge data for each water year (October 1 – September 30) were used to calculate annual average flow rates. Trends were determined by evaluating the probability that the slope of the average annual flow versus year, as determined through simple linear regression, was significantly different than zero ($p<0.05$).

The significance of the Pearson's correlation coefficient was determined by calculating the probability that the correlation was different than zero based on the value of the correlation and the sample size. A significant correlation does not indicate a strong correlation.

Summary

Of the 14 locations analyzed, only one showed a significant change in overall annual flow. All other results were not significant ($p>0.10$). Annual Average Flow rates are informative when used in combination with other hydrologic indicators such as summer low flows and indicator of flow timing.

Literature Cited

Bosch, J. M. and J. D. Hewlett. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapo-transpiration. Journal of Hydrology 55:3-23.

Cuo, L., D. P. Lettenmaier, M. Alberti, and J. E. Richey. 2009. Effects of a century of land cover and climate change on the hydrology of the Puget Sound basin. Hydrological Processes 23:907-933.

Elsner, M. M., L. Cuo, N. Voisin, J. S. Deems, A. F. Hamlet, J. A. Vano, K. E. Mickelson, S.-Y. Lee, and D. P. Lettenmaier. 2009. Implications of 21st Century Climate Change for the

Hydrology of Washington State. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Hamlet, A. F. and D. P. Lettenmaier. 2007. Effects of 20th century warming and climate variability on flood risk in the western U.S. Water Resources Research 43.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2005. Effects of temperature and precipitation variability on snowpack trends in the western United States. Journal of Climate 18:4545-4561.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2007. Twentieth-century trends in runoff, evapotranspiration, and soil moisture in the western United States. Journal of Climate 20:1468-1486.

Jain, S., M. Hoerling, and J. Eischeid. 2005. Decreasing reliability and increasing synchroneity of western North American streamflow. Journal of Climate 18:613-618.

Luce, C. H. and Z. A. Holden. 2009. Declining annual streamflow distributions in the Pacific Northwest United States, 1948-2006. Geophysical Research Letters 36.

Mote, P. W., A. F. Hamlet, M. P. Clark, and D. P. Lettenmaier. 2005. Declining mountain snowpack in western north America. Bulletin of the American Meteorological Society 86:39-+.

Pagano, T. and D. Garen. 2005. A recent increase in western US streamflow variability and persistence. Journal of Hydrometeorology 6:173-179.

Puget Sound Partnership. 2009. Ecosystem Status and Trends. Water Quantity., Puget Sound Partnership.

Simmons, D. L. and R. J. Reynolds. 1982. Effects of urbanizatsion on base-flow of selected south-shore streams, Long Island, New York. Water Resources Bulletin 18:797-805.

United States Geological Survey. 2010a. Provisional data for Washington.

United States Geological Survey. 2010b. USGS Washington Water Science Center.

Average Daily Flow

Background

Streamflow patterns in Puget Sound rivers and streams are classified into three hydrologic regimes: snowmelt dominated, rainfall dominated, and transitional (Stewart et al. 2005, Beechie et al. 2006, Elsner et al. 2009). Generally, in snowmelt-dominated rivers, a significant proportion of winter precipitation is stored as snowpack, resulting in low winter flows with peak flows during the spring snowmelt period from April through July. Rainfall-dominated rivers experience peak flow during the winter months as the majority of precipitation falls as rain. Transitional rivers experience both winter and spring peak flows resulting from winter precipitation and spring snowmelt. Hydrologic flow regimes in Puget Sound rivers have been altered through the construction of dams for flood control or power generation, or by changes in land cover and climate. Alteration of historical flow patterns can cause ecological harm and disrupt supply (Poff et al. 1997, Wiley and Palmer 2008).

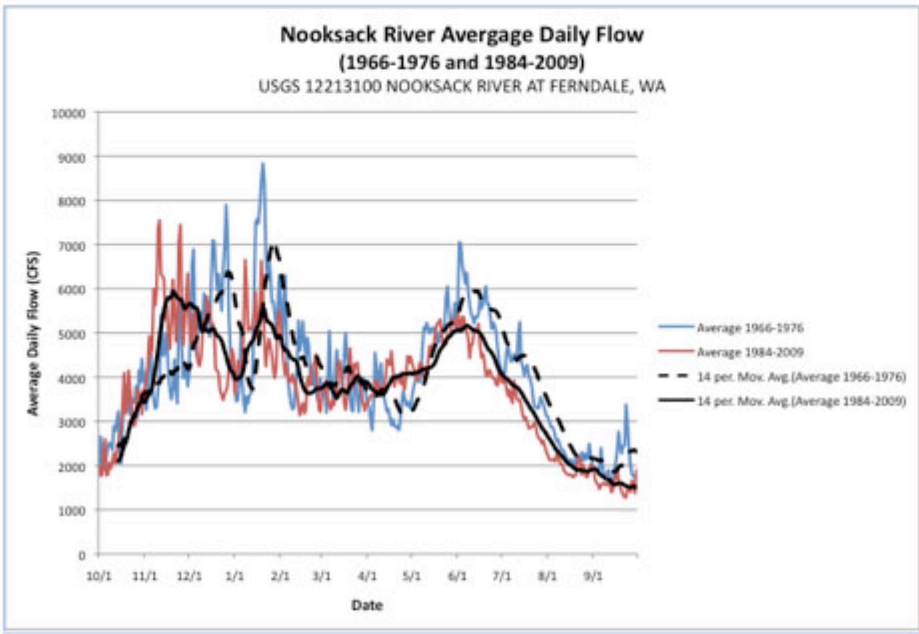
Barnett et al. (2008) utilized a multivariate analysis to evaluate simultaneous changes in average winter temperature, snow pack, and runoff timing in the Western United States (including the Washington Cascades) for the period from 1950 – 1999. They found significant increasing trends in winter temperature and decreasing trends in snow pack and runoff timing (indicating earlier snowmelt). In order to distinguish natural variation from anthropogenic forcing, they evaluated the observations against two separate climate models and found that the hydrologic changes were both detectable and attributable to anthropogenic forcings.

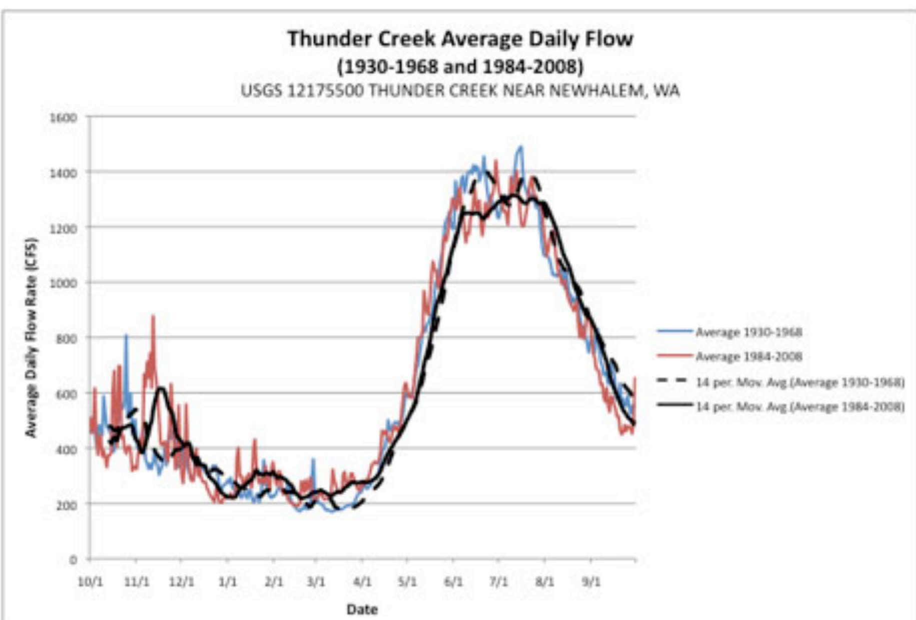
Stewart et al. (2004) investigated historic (1948-2000) and future streamflow timing in snowmelt dominated rivers and streams in the Western United States. They found significant trends towards earlier runoff in many rivers and streams in the Pacific Northwest. Utilizing a ‘business-as-usual’ emissions scenario with a Parallel Climate Model, they predicted continuation of this trend, due largely to increased winter and spring temperatures but not changes in precipitation. In a companion study they further analyzed the trends in streamflow timing with variations of the PDO (Stewart et al. 2005). While streamflow timing was partially controlled by the PDO there remained a substantial portion of the variation in timing that was explained by a longer-term warming trend in spring temperatures.

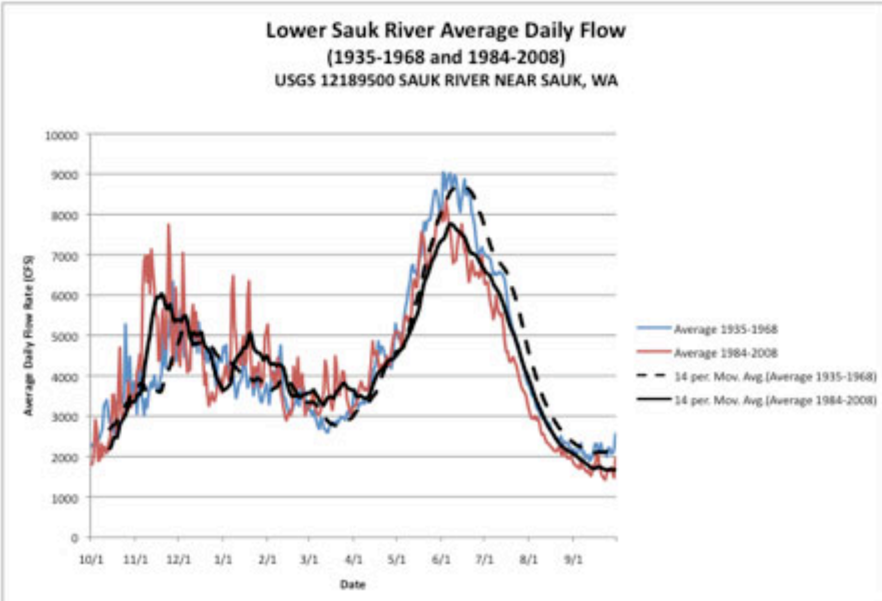
The Climate Impact Group at the University of Washington performed The Washington Climate Change Impact Assessment. The assessment included analyses of hydrology and water resource management utilizing results from 20 global climate models and two emissions scenarios from the IPCC Special Report on Emissions Scenarios (A1B and B1) to evaluate projected changes in spring snowpack and runoff (Elsner et al. 2009). For the rivers in the Puget Sound basin, they found a dramatic decrease in spring snowpack with there being almost no April 1 snowpack by 2080. Change in snowpack was correlated with a predicted change in river hydrography, from transition- or snow-rain dominated, to rain dominated patterns. There was little predicted change in annual precipitation.

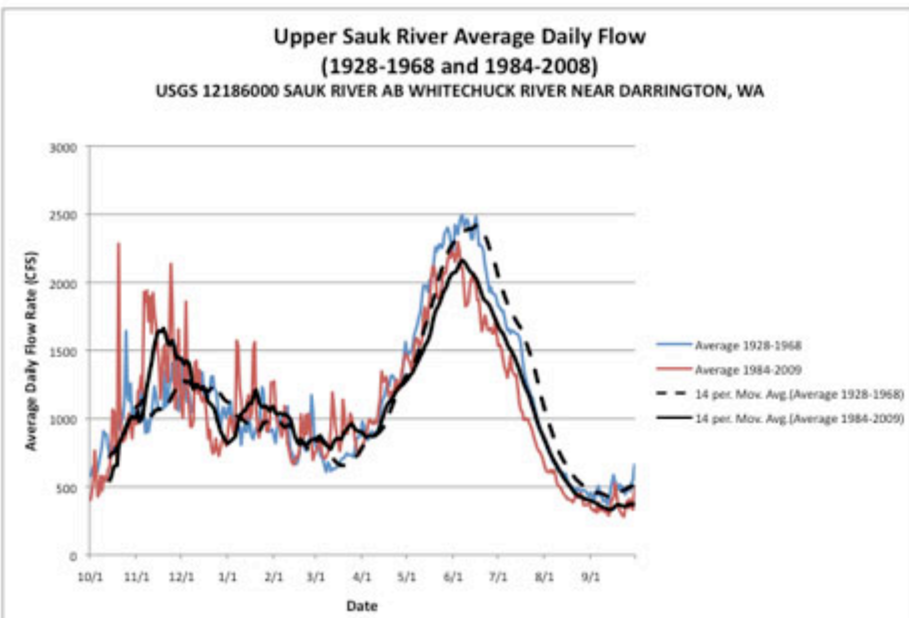
Status and Trends

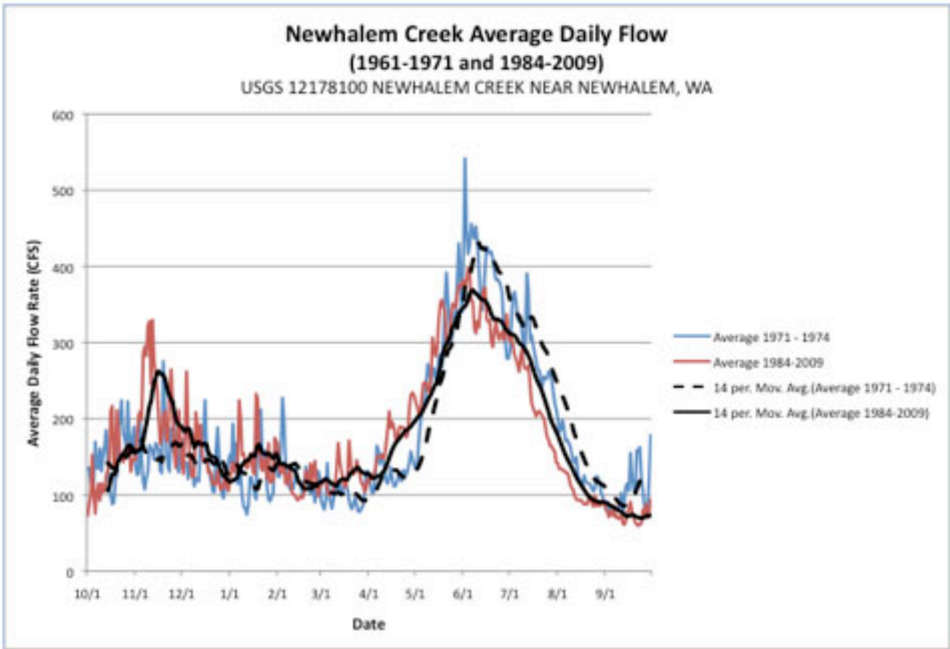
River hydrographs showing average annual daily flow from the initiative of observation through 1968, and from 1984 through 2009 are presented in Figure 1. Much of the warming trend observed in the Pacific Northwest has occurred since 1975 (Hamlet and Lettenmaier 2007). Comparing the streamflow patterns before and after this period could indicate effects of climate change.

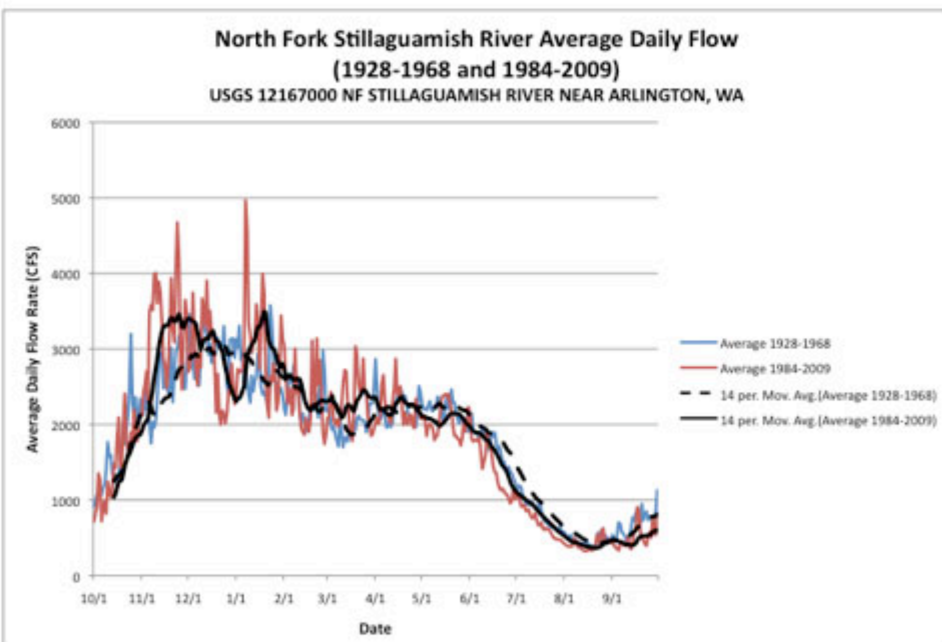


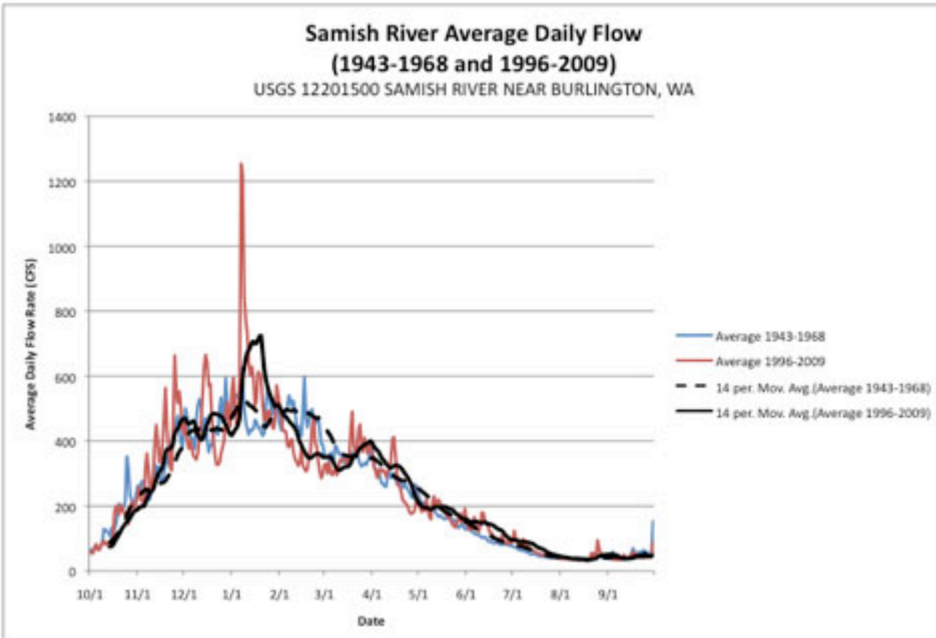



Lower Sauk River Average Daily Flow
(1935-1968 and 1984-2008)
USGS 12189500 SAUK RIVER NEAR SAUK, WA
Average Daily Flow Rate (CFS)
Date
Average 1935-1968
Average 1984-2008
14 per. Mov. Avg.(Average 1935-1968)
14 per. Mov. Avg.(Average 1984-2008)

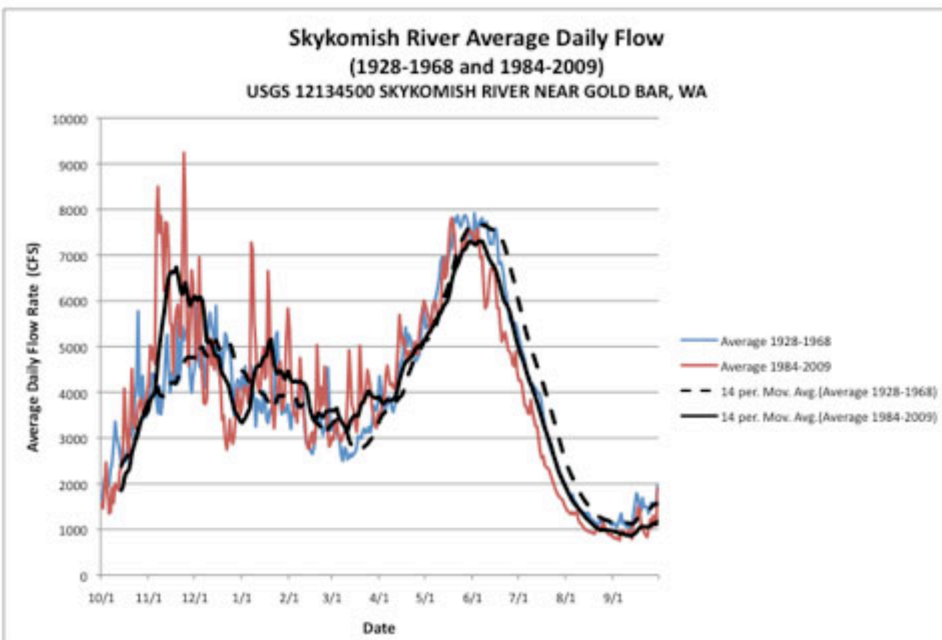

Upper Sauk River Average Daily Flow
(1928-1968 and 1984-2008)
USGS 12186000 SAUK RIVER AB WHITECHUCK RIVER NEAR DARRINGTON, WA
Average Daily Flow Rate (CFS)
Date
Average 1928-1968
Average 1984-2009
14 per. Mov. Avg.(Average 1928-1968)
14 per. Mov. Avg.(Average 1984-2009)


Newhalem Creek Average Daily Flow
(1961-1971 and 1984-2009)
USGS 12178100 NEWHALEM CREEK NEAR NEWHALEM, WA
Average Daily Flow Rate (CFS)
600
500
400
300
200
100
0
10/1 11/1 12/1 1/1 2/1 3/1 4/1 5/1 6/1 7/1 8/1 9/1
Date
Average 1971 - 1974
Average 1984-2009
14 per. Mov. Avg.(Average 1971 - 1974)
14 per. Mov. Avg.(Average 1984-2009)

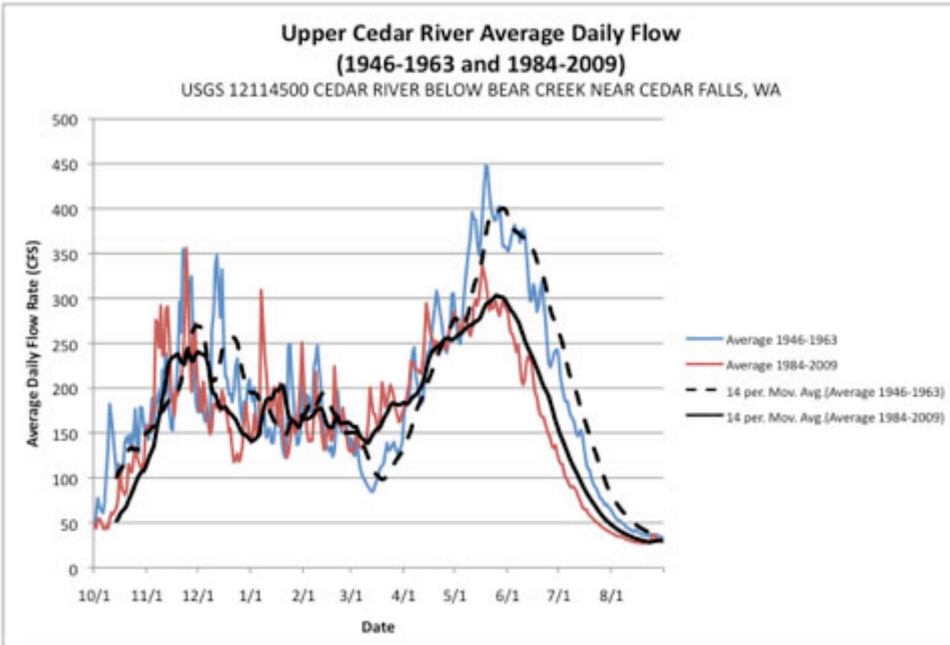

North Fork Stillaguamish River Average Daily Flow
(1928-1968 and 1984-2009)
USGS 12167000 NF STILLAGUAMISH RIVER NEAR ARLINGTON, WA
Average Daily Flow Rate (CFS)
6000
5000
4000
3000
2000
1000
0
10/1 11/1 12/1 1/1 2/1 3/1 4/1 5/1 6/1 7/1 8/1 9/1
Date
Average 1928-1968
Average 1984-2009
14 per. Mov. Avg.(Average 1928-1968)
14 per. Mov. Avg.(Average 1984-2009)

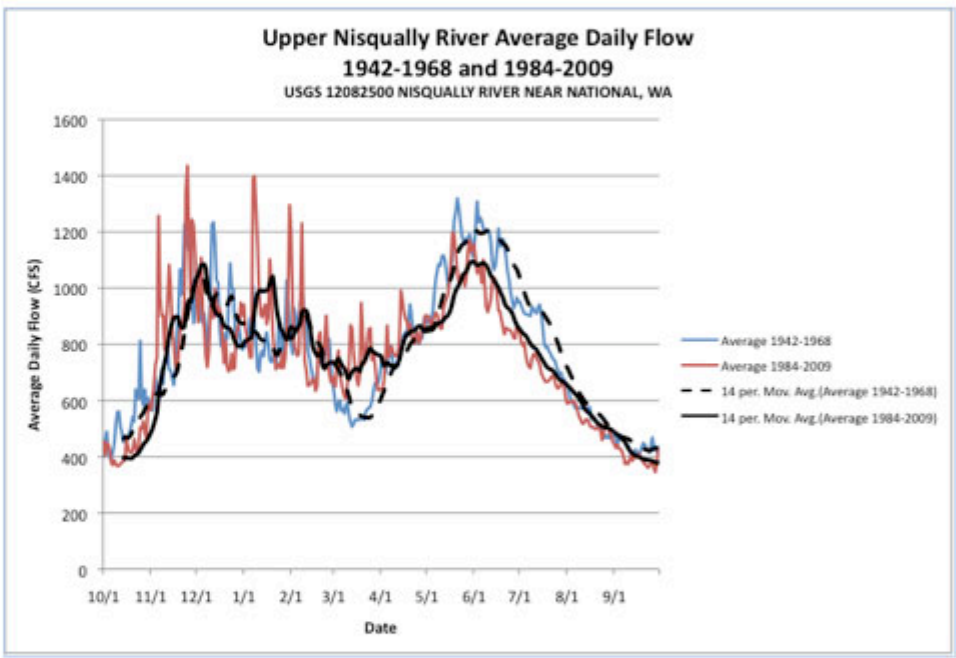


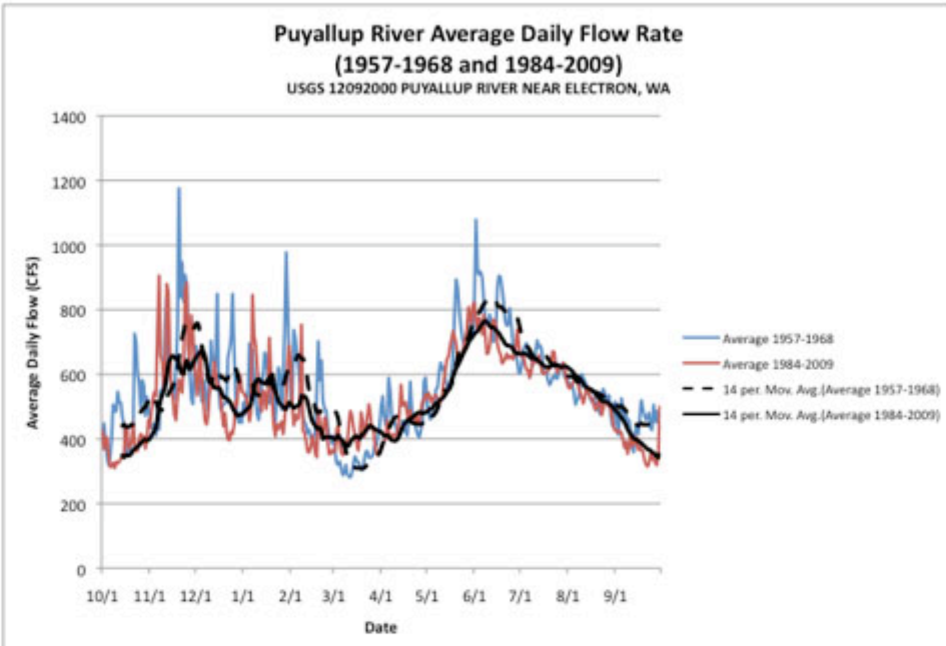
Samish River Average Daily Flow
(1943-1968 and 1996-2009)
USGS 12201500 SAMISH RIVER NEAR BURLINGTON, WA
Average Daily Flow Rate (CFS)
Date
Average 1943-1968
Average 1996-2009
14 per. Mov. Avg.(Average 1943-1968)
14 per. Mov. Avg.(Average 1996-2009)



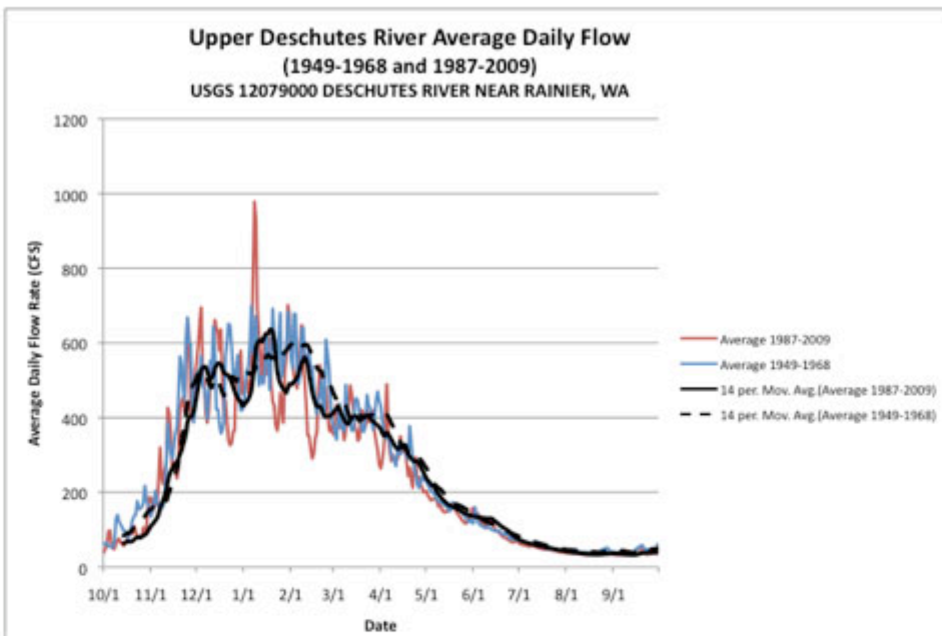
Skykomish River Average Daily Flow
(1928-1968 and 1984-2009)
USGS 12134500 SKYKOMISH RIVER NEAR GOLD BAR, WA
Average Daily Flow Rate (CFS)
Date
Average 1928-1968
Average 1984-2009
14 per. Mov. Avg.(Average 1928-1968)
14 per. Mov. Avg.(Average 1984-2009)


Upper Cedar River Average Daily Flow
(1946-1963 and 1984-2009)
USGS 12114500 CEDAR RIVER BELOW BEAR CREEK NEAR CEDAR FALLS, WA
Average Daily Flow Rate (CFS)
500
450
400
350
300
250
200
150
100
50
0
10/1
11/1
12/1
1/1
2/1
3/1
4/1
5/1
6/1
7/1
8/1
Date
Average 1946-1963
Average 1984-2009
14 per. Mov. Avg.(Average 1946-1963)
14 per. Mov. Avg.(Average 1984-2009)

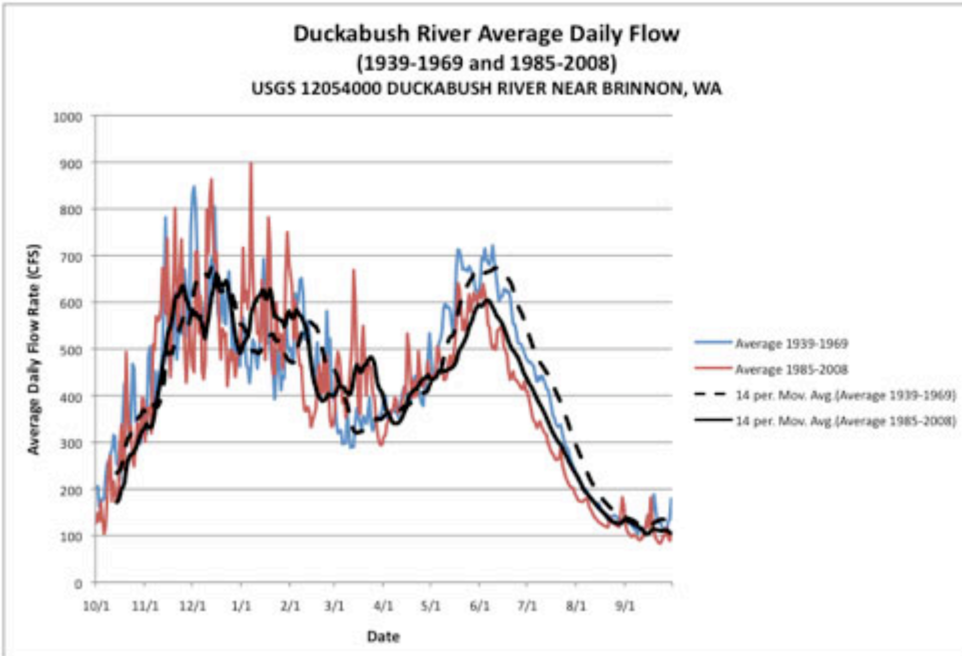

Upper Nisqually River Average Daily Flow
1942-1968 and 1984-2009
USGS 12082500 NISQUALLY RIVER NEAR NATIONAL, WA
Average Daily Flow (CFS)
1600
1400
1200
1000
800
600
400
200
0
10/1
11/1
12/1
1/1
2/1
3/1
4/1
5/1
6/1
7/1
8/1
9/1
Date
Average 1942-1968
Average 1984-2009
14 per. Mov. Avg.(Average 1942-1968)
14 per. Mov. Avg.(Average 1984-2009)



Puyallup River Average Daily Flow Rate
(1957-1968 and 1984-2009)
USGS 12092000 PUYALLUP RIVER NEAR ELECTRON, WA
Average Daily Flow (CFS)
1400
1200
1000
800
600
400
200
0
10/1 11/1 12/1 1/1 2/1 3/1 4/1 5/1 6/1 7/1 8/1 9/1
Date
Average 1957-1968
Average 1984-2009
14 per. Mov. Avg.(Average 1957-1968)
14 per. Mov. Avg.(Average 1984-2009)



Upper Deschutes River Average Daily Flow
(1949-1968 and 1987-2009)
USGS 12079000 DESCHUTES RIVER NEAR RAINIER, WA
Average Daily Flow Rate (CFS)
1200
1000
800
600
400
200
0
10/1 11/1 12/1 1/1 2/1 3/1 4/1 5/1 6/1 7/1 8/1 9/1
Date
Average 1987-2009
Average 1949-1968
14 per. Mov. Avg.(Average 1987-2009)
14 per. Mov. Avg.(Average 1949-1968)



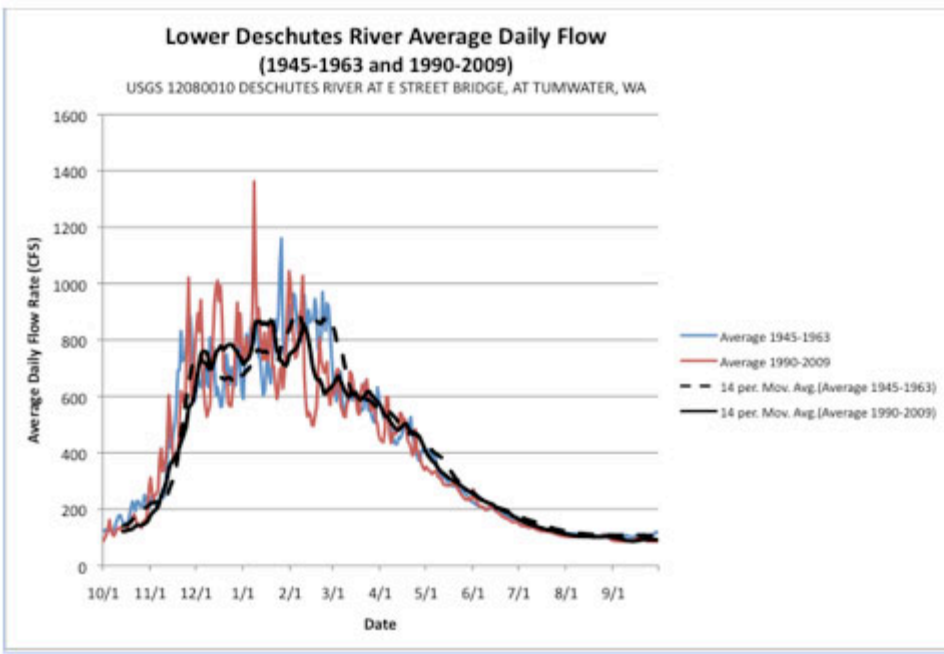


Figure 1. Average daily flow shown for historic (pre 1970's) and recent (mid-1980s to present) time period for 14 Puget Sound rivers. The time period varies slightly between river systems based on availability of data. Colored lines show average daily flow averaged over time period indicated in each of the chart title. Dark lines are 14-day smoothed averages for historic (dashed) or recent (solid) time periods. Data taken from United States Geological Service.

There is considerable variation even in averaged data which makes the detection of long-term trends problematic. However, the following generalities emerge. First, there has been little change in hydrologic patterns in rainfall-dominated rivers (Samish, Stillaguamish, and Deschutes) or in the snowmelt-dominated river (Thunder Creek). It is possible that consistent glacier melt contributed to the stable patterns in the latter river. However, there was an observable decline in spring peak flows in all of the transitional rivers (Nooksack, Sauk, Newhalem Creek, Skykomish, Upper Cedar, Upper Puyallup, Upper Nisqually, and Duckabush). Moreover, there appears to be a decline in the magnitude of the summer 7-day average low flows.

Uncertainties

The analysis presented above was derived from data in the public domain. Hydrographs were created utilizing average daily discharge data from USGS stations located in the Puget Sound region (United States Geological Survey 2010b). The datasets include qualification codes indicating whether data are provisional or have been approved (United States Geological Survey 2010a). We avoided using provisional data in this analysis, and we omitted data from gauging stations for which advisory notes warning against unreliable data quality had been posted.

The analysis in this section is qualitative and intended to illustrate potential changes in streamflow patters over time. Consequently, statistical significance was not determined. Specific streamflow measures, such as annual 7-day average low flow, or centroid of flow timing, are quantitative measures that can be evaluated statistically and are presented elsewhere in this document.

Summary

There is some evidence for changes in transitional river systems over time, indicated primarily as decreasing magnitude of the spring snowmelt peak flows. This is consistent with published predictions for the western North America. There also appears to be a decrease in the magnitude of summer low flows in transitional river systems. There was less evidence for change in daily flow patterns for rainfall-dominated or snowmelt-dominated river systems. Because of variation in hydrologic alteration, particularly between rivers or streams of differing classifications, combining streamflow information across multiple streams to evaluate general status and trends may not be appropriate and results should be interpreted with caution.

Literature Cited

Barnett, T. P., D. W. Pierce, H. G. Hidalgo, C. Bonfils, B. D. Santer, T. Das, G. Bala, A. W. Wood, T. Nozawa, A. A. Mirin, D. R. Cayan, and M. D. Dettinger. 2008. Human-induced changes in the hydrology of the western United States. Science 319:1080-1083.

Beechie, T., E. Buhle, M. Ruckelshaus, A. Fullerton, and L. Holsinger. 2006. Hydrologic regime and the conservation of salmon life history diversity. Biological Conservation 130:560-572.

Elsner, M. M., L. Cuo, N. Voisin, J. S. Deems, A. F. Hamlet, J. A. Vano, K. E. Mickelson, S.-Y. Lee, and D. P. Lettenmaier. 2009. Implications of 21st Century Climate Change for the Hydrology of Washington State. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Hamlet, A. F. and D. P. Lettenmaier. 2007. Effects of 20th century warming and climate variability on flood risk in the western U.S. Water Resources Research 43.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. Bioscience 47:769-784.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2004. Changes in snowmelt runoff timing in western North America under a 'business as usual' climate change scenario. Climatic Change 62:217-232.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate 18:1136-1155.

United States Geological Survey. 2010a. Provisional data for Washington.

United States Geological Survey. 2010b. USGS Washington Water Science Center.

Wiley, M. W. and R. N. Palmer. 2008. Estimating the impacts and uncertainty of climate change on a municipal water supply system. Journal of Water Resources Planning and Management-Asce 134:239-246.

Seven-day Average Low Flow

Background

The hydrologic regime of rivers and streams in the Puget Sound is characterized by peak flows during the winter as a result of heavy precipitation, or during the spring due to snowmelt runoff. Base flows during the summer are low, consisting mainly of groundwater discharge. Base flows can be affected by climate change, urbanization, or groundwater withdrawals. Summer base flow levels are important ecologically because they can define or limit the availability of habitats. Summer base flow levels are important to water resource managers because low flows often coincide with peak consumption.

Climate change is expected to alter river hydrology in the Puget Sound basin. Observed and predicted increases in winter temperatures could result in more precipitation falling as rain instead of snow, earlier snowmelt timing, earlier streamflow timing, and lower summer flows (Mote et al. 1999, Mote et al. 2003). Several studies have evaluated the impacts of climate change on spring snowpack in the Pacific Northwest, with the conclusion that decreasing spring snowpack may result in lower summer flows. Long-term decline in snowpack in the Pacific Northwest was found to correlate largely with increasing temperatures, but not precipitation (Mote 2003). Follow-on studies with a Variable Infiltration Capacity model were performed to discern long term trends in spring snowpack from temperature and precipitation variability (Hamlet et al. 2005, Mote et al. 2005). Results suggested that long-term downward trends in spring snowpack were associated with widespread warming. Trends in snowpack associated with precipitation were largely controlled by decadal oscillations. Multiple regression analysis indicated that climatic oscillations accounted for approximately 10-60% of the trends in spring snowpack, depending on the time series examined (Mote 2006), leading the authors to conclude that the primary factor driving declining snowpack in the Washington Cascades was rising temperatures. The long-term snowpack trends were unrelated to the variability caused by Pacific oscillations.

Casola et al. (2009) investigated the potential impacts of climate change on snowpack by combining future temperature predictions with the estimated temperature sensitivity of spring snowpack. Analysis of historic and projected temperature data indicated that snowpack reductions over the past 30 years ranged from 8%-16% while future temperature change would result in an 11%-21% reduction in spring snowpack by 2050.

Stewart et al. (2005) evaluated the monthly fractional flow in snowmelt-dominated river systems in the Western United States and found an increasing fraction of flow occurring in March, corresponding with a decreasing fraction in June. Changes in streamflow pattern were associated with long-term increases in spring and winter temperatures, which spanned the decadal-scale Pacific climate oscillations. Barnett et al. (2008) utilized a multivariate analysis to evaluate the simultaneous changes in average winter temperature, snow pack, and runoff timing in the Western United States (including the Washington Cascades) for the period from 1950 – 1999 They found significant increasing trends in winter temperature, and decreasing trends in snow pack and runoff timing (indicating earlier snowmelt) associated with anthropogenic forcings.

The Climate Impacts Group utilized results from 20 global climate models and two emissions scenarios from the IPCC Special Report on Emissions Scenarios (A1B and B1) to evaluate projected changes in spring snowpack and runoff (Elsner et al. 2009). For the rivers in the Puget Sound basin they projected a dramatic decrease in spring snowpack with almost no April 1 snowpack by 2080. The climate change-related alterations in spring snowpack and streamflow timing are expected to result in lower summer flows.

Land use alterations can also result in lower summer flows. Urbanization and development are associated with an increase in impervious surface resulting in higher runoff fractions and lower infiltration (Burges et al. 1998). Reduced infiltration can lead to lower base flows, although this effect can be somewhat offset by a reduction in evapo-transpiration from the clearing of trees (Cuo et al. 2008). The construction of storm drain systems has been implicated as a primary factor in the reduction a base flows (Simmons and Reynolds 1982).

Cuo et al. (2009) utilized a Distributed Hydrology-Soil-Vegetation Model in order to determine the relative effects of land cover and temperature change on flow patterns in Puget Sound streams. They found that the relative importance of temperature and land cover differed between the upland and lowland basins. In the lowland basins land cover changes were more important and generally resulted in higher peak flows and lower summer flows, primarily due to increased runoff. Both land use change and climate effects were important in the upland basins. Climate change had the largest impact in the transitional zones and resulted in higher winter flows, earlier spring peak flows, and lower summer flows. A similar modeling study of a basin located in the Portland, OR metropolitan area, using a single climate change simulation combined with a ArcView Soil and Water Assessment Tool, predicted an increase in overall flow, but a decrease in summer baseflow, by 2040 (Franczyk and Chang 2009).

Monitoring trends and predicting potential future alterations in streamflow patterns is important for water resource managers to ensure sufficient supply to meet demand (Snover et al. 2003, Milly et al. 2008). In the Pacific Northwest, summer low flows define the crucial period of water use and availability, and define system yield capacity. Wiley and Palmer utilized a three-stage modeling approach to evaluate the impacts of climate change on the water supply system for Seattle metropolitan region (Wiley and Palmer 2008). They predicted a decline of 6% per decade in July-September reservoir inflows resulting in a loss of available water in the system of approximately 56,000 acre-feet by 2040. Climate-related changes may reduce overall system yield.

Vano et al. (2009) expanded the analysis to include the Everett and Tacoma water supply systems. They predicted decreased summer reservoir inflows and storage for all three systems. System reliability, however, remained relatively strong assuming current demand.

Summer low flows in streams and rivers may be ecologically important. A substantial body of literature describes the potential deleterious impacts of low summer flows on fish survival (see Crozier et al. 2008, Palmer et al. 2009 and references therein). Potential negative biological impacts of low summer flows include high water temperatures, stranding, low dissolved oxygen, crowding, and disease. Although the strength of salmon runs has been shown to be positively and significantly correlated to summer stream flow in Puget Sound rivers, the actual causative mechanism is unclear due to complicated and interrelated variations between flow, temperature, habitat, and other variables (Mathews and Olson 1980). Rand et al. (2006) evaluated the potential effects of reduced flow and increased water temperature on upriver migration of Pacific salmon in the Frasier River. Lower discharge volumes during the migration period increased survival by decreasing energy requirements of the migrating salmon (making it easier to swim upstream) leading to a stronger pre-spawn population. Higher water temperatures, however, have been shown to increase metabolic rates and increase energy requirements. Presumably, within some range, the energetic benefits of decreased flow will compensate for costs from higher temperatures, yielding no net effect.

Scheuerell et al. (2006) used summer stream temperatures, which are predicted to increase with decreased flow, as a negative factor in survival of Chinook salmon in an effort to model salmon survival according to changes in various environmental conditions. Battin et al. (2007) predicted that Chinook salmon spawner capacity was proportional to minimum discharge during the spawning period; reductions in flow would result in reductions in spawning capacity due to habitat limitations. Low flows are also important for juvenile Coho due to space and food limitations, while low flows may be associated with temperature limitations in other areas (Ebersole et al. 2009). Trout survival and growth have been shown to be negatively associated with low stream discharge (Harvey et al. 2006, Berger and Gresswell 2009).

There remains substantial uncertainty in the predicted changes, related not only to climate change, but also to biological response and potential for adaptation among various species, particularly salmonids(Crozier et al. 2008, Schindler et al. 2008). Biological responses are likely to vary according to the specific stream and basin.

Status and trends

Summer 7-day average low flow is the metric chosen to represent low stream flow conditions. It is widely used and not susceptible to temporary upstream flow changes than may affect one-day low flow calculations (Riggs 1985). Annual values for 7-day average low flow were calculated using gauge data from 14 different locations on unregulated rivers within the Puget Sound, in order to evaluate the status and trends of low flows within the region (Table 1). Data from seven rivers indicated a significantly decreasing trend in 7-day average low flow for the time period on record ($p<0.05$). Data from three other rivers indicated decreasing trends in 7-day average low flow, although with a slightly higher degree of statistical uncertainty ($p<0.10$). Four rivers showed no significant trends in annual 7-day average low flow. Notably, no river system showed significantly increasing trends in annual 7-day average low flow. The average change for the rivers with significant trends in annual 7-day average low flow was -4.4% per decade.

Table 1. Average 7-day Low Flow for the time period of record, the annual rate of change of 7-day low flow, and the probability that the trend is significantly different than zero for selected unregulated rivers and streams in the Puget Sound basin.

			7-DAY AVERAGE LOW FLOW		
River	Data Years		Average Low Flow (CFS)	Annual Change (ΔCFS/Year)	p (change≠0)
WRIA 1 – Nooksack					
Nooksack USGS 12213100	1966-2009		1020	-6.3±3.6	0.09
WRIA 3/4 – Upper-Lower Skagit and Samish					
Lower Sauk USGS 12189500	1936-2009		1281	-3.5±2.0	0.08
Upper Sauk USGS 12186000	1929-2009		231	-0.6±0.4	0.09
Thunder USGS 12175500	1931-2009		225	-0.2±0.4	0.55
Newhalem USGS 12178100	1962-2009		45	-0.3±0.1	0.03
Samish USGS 12201500	1945-1970 1996-2009		27	0.01±0.04	0.77
WRIA 5 - Stillaguamish					
Stillaguamish USGS 12167000	1929-2009		255	-0.6±0.4	0.18
WRIA 7 – Snohomish					
Skykomish USGS 12134500	1929-2009		655	-2.2±1.1	0.05
WRIA 8 – Cedar/Sammamish					
Cedar USGS 12114500	1947-2009		22	-0.1±0.04	0.002
WRIA 10 – Puyallup/White					
Puyallup USGS 12092000	1957-2009		216	-1.2±0.5	0.03
WRIA 11 - Nisqually					
Nisqually USGS 12082500	1942-2009		263	-0.7±0.5	0.14
WRIA 13 - Deschutes					
Lower Deschutes USGS 12080010	1946-1963 1990-2009		83	-0.4±0.1	0.001
Upper Deschutes USGS 12079000	1950-2009		29	-0.1±0.04	0.0002
WRIA 16 – Skokomish/Dosewalips					
Duckabush USGS 12054000	1939-2009		72	-0.3±0.1	0.04

There were no consistently strong correlations between the annual 7-day average low flow values for the rivers within WRIA 3/4 (Table 2). Calculated annual 7-day average low flow values from Thunder Creek and the Samish River generally correlate weakly with the other rivers within the group used for comparison. Thunder Creek can be classified as a snowmelt-dominated river. The Samish River is a rainfall-dominated river. The other rivers within the group are all transition rivers. It is possible that the different hydrologic regimes partially explain the lack of correlations in low flow.

Table 2. Pearson's correlation coefficient for annual 7-day average low flow for rivers within WRIA 3/4.

	Lower Sauk	Upper Sauk	Thunder	Cascade	Newhalem	Samish
Lower Sauk		0.84	0.54	0.91	0.73	0.34

Upper Sauk	0.44	0.77	0.80	0.49
Thunder		0.72	0.42	-0.11a
Cascade			0.80	0.23a
Newhalem				0.54

Notes: a. Pearson's r not significantly different than 0 (P>0.05)

Uncertainties

The analysis presented above was derived from data in the public domain. The values and trends for 7-day average low flow were calculated from average daily discharge data from fourteen USGS station located in the Puget Sound region (United States Geological Survey 2010b). The datasets include qualification codes indicating whether data are provisional or have been approved (United States Geological Survey 2010a). We avoided using provisional data in this analysis, and we omitted data from gauging stations for which advisory notes warning against unreliable data quality had been posted.

The 7-day low flow values were calculated for the period from June 1 – November 1; this time period was chosen to avoid the potential capture of winter low flows in the snowmelt-dominated river system (e.g., Thunder Creek). Trends were determined by calculating the slope of the annual 7-day low flow versus year using simple linear regression. Significance was determined by applying the Student's t-test to determine the probability of the slope being significantly different than zero (P<0.05).

The significance of the Pearson's correlation coefficient was determined by estimating the probability that the correlation was different than zero based on the value of the correlation and the sample size. A significant correlation does not indicate a strong correlation.

Summary

Analysis of streamflow data revealed decreasing trends in 7-day average low flow values for seven of 14 guaging stations. Among the remaining stations, none showed significant increasing trends. Substantial inter-annual variation in low flow was evident. Annual 7-day average low flows among the river systems in WRIA 3/4 showed no consistent correlation. The weakest correlations were between the snowmelt-dominated (Thunder Creek), the rainfall-dominated (Samish River) and the remaining river systems. Seven-day average low flow could be a useful indicator of changing conditions in these watersheds.

Literature Cited

Barnett, T. P., D. W. Pierce, H. G. Hidalgo, C. Bonfils, B. D. Santer, T. Das, G. Bala, A. W. Wood, T. Nozawa, A. A. Mirin, D. R. Cayan, and M. D. Dettinger. 2008. Human-induced changes in the hydrology of the western United States. Science 319:1080-1083.

Battin, J., M. W. Wiley, M. H. Ruckelshaus, R. N. Palmer, E. Korb, K. K. Bartz, and H. Imaki. 2007. Projected impacts of climate change on salmon habitat restoration. Proceedings of the National Academy of Sciences of the United States of America 104:6720-6725.

Berger, A. M., and R. E. Gresswell. 2009. Factors influencing coastal cutthroat trout (Oncorhynchus clarkii clarkii) seasonal survival rates: a spatially continuous approach within stream networks. Canadian Journal of Fisheries and Aquatic Sciences 66:613-632.

Burges, S. J., M. S. Wigmosta, and J. M. Meena. 1998. Hydrological Effects of Land-Use Change in a Zero-Order Catchment. Journal of Hydrologic Engineering 3:86-97.

Casola, J. H., L. Cuo, B. Livneh, D. P. Lettenmaier, M. T. Stoelinga, P. W. Mote, and J. M. Wallace. 2009. Assessing the impacts of global warming on snowpack in the Washington Cascades. Journal of Climate 22:2758-2772.

Crozier, L. G., A. P. Hendry, P. W. Lawson, T. P. Quinn, N. J. Mantua, J. Battin, R. G. Shaw, and R. B. Huey. 2008. Potential responses to climate change in organisms with complex life histories: evolution and plasticity in Pacific salmon. Evolutionary Applications 1:252-270.

Cuo, L., D. P. Lettenmaier, M. Alberti, and J. E. Richey. 2009. Effects of a century of land cover and climate change on the hydrology of the Puget Sound basin. Hydrological Processes 23:907-933.

Cuo, L., D. P. Lettenmaier, B. V. Mattheussen, P. Storck, and M. Wiley. 2008. Hydrologic prediction for urban watersheds with the Distributed Hydrology-Soil-Vegetation Model. Hydrological Processes 22:4205-4213.

Ebersole, J. L., M. E. Colvin, P. J. Wigington, S. G. Leibowitz, J. P. Baker, M. R. Church, J. E. Compton, and M. A. Cairns. 2009. Hierarchical Modeling of Late-Summer Weight and Summer Abundance of Juvenile Coho Salmon across a Stream Network. Transactions of the American Fisheries Society 138:1138-1156.

Elsner, M. M., L. Cuo, N. Voisin, J. S. Deems, A. F. Hamlet, J. A. Vano, K. E. Mickelson, S.-Y. Lee, and D. P. Lettenmaier. 2009. Implications of 21st Century Climate Change for the Hydrology of Washington State. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Franczyk, J., and H. Chang. 2009. The effects of climate change and urbanization on the runoff of the Rock Creek basin in the Portland metropolitan area, Oregon, USA. Hydrological Processes 23:805-815.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2005. Effects of temperature and precipitation variability on snowpack trends in the western United States. Journal of Climate 18:4545-4561.

Harvey, B. C., R. J. Nakamoto, and J. L. White. 2006. Reduced streamflow lowers dry-season growth of rainbow trout in a small stream. Transactions of the American Fisheries Society 135:998-1005.

Mathews, S. B., and F. W. Olson. 1980. Factors affecting Puget Sound coho (Oncorhynchus kisutch) runs. Canadian Journal of Fisheries and Aquatic Sciences 37:1373-1378.

Milly, P. C. D., J. Betancourt, M. Falkenmark, R. M. Hirsch, Z. W. Kundzewicz, D. P. Lettenmaier, and R. J. Stouffer. 2008. Climate change - Stationarity is dead: Whither water management? Science 319:573-574.

Mote, P., D. Canning, D. Fluharty, R. Francis, J. Franklin, A. Hamlet, M. Hershman, M. Holmberg, K. G. Ideker, W. Keeton, D. Lettenmaier, R. Leung, N. Mantua, E. Miles, B. Noble, H. Parandvash, D. W. Peterson, A. Snover, and S. Willard. 1999. Impacts of climate variability and change inthe Pacific Northwest. The JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Mote, P. W. 2003. Trends in snow water equivalent in the Pacific Northwest and their climatic causes. Geophysical Research Letters 30.

Mote, P. W. 2006. Climate-driven variability and trends in mountain snowpack in western North America. Journal of Climate 19:6209-6220.

Mote, P. W., A. F. Hamlet, M. P. Clark, and D. P. Lettenmaier. 2005. Declining mountain snowpack in western north America. Bulletin of the American Meteorological Society 86:39-+.

Mote, P. W., E. Parson, A. F. Hamlet, W. S. Keeton, D. Lettenmaier, N. Mantua, E. L. Miles, D. Peterson, D. L. Peterson, R. Slaughter, and A. K. Snover. 2003. Preparing for climatic change: The water, salmon, and forests of the Pacific Northwest. Climatic Change 61:45-88.

Palmer, M. A., D. P. Lettenmaier, N. L. Poff, S. L. Postel, B. Richter, and R. Warner. 2009. Climate Change and River Ecosystems: Protection and Adaptation Options. Environmental Management 44:1053-1068.

Rand, P. S., S. G. Hinch, J. Morrison, M. G. G. Foreman, M. J. MacNutt, J. S. Macdonald, M. C. Healey, A. P. Farrell, and D. A. Higgs. 2006. Effects of river discharge, temperature, and future climates on energetics and mortality of adult migrating Fraser River sockeye salmon. Transactions of the American Fisheries Society 135:655-667.

Riggs, H. C. 1985. Streamflow characteristics. Elsevier, Amsterdam; New York.

Scheuerell, M. D., R. Hilborn, M. H. Ruckelshaus, K. K. Bartz, K. M. Lagueux, A. D. Haas, and K. Rawson. 2006. The Shiraz model: a tool for incorporating anthropogenic effects and fish-habitat relationships in conservation planning. Canadian Journal of Fisheries and Aquatic Sciences 63:1596-1607.

Schindler, D. E., X. Augerot, E. Fleishman, N. J. Mantua, B. Riddell, M. Ruckelshaus, J. Seeb, and M. Webster. 2008. Climate Change, Ecosystem Impacts, and Management for Pacific Salmon. Fisheries 33:502-506.

Simmons, D. L., and R. J. Reynolds. 1982. Effects of urganization on base-flow of selected South-shore streams, Long Island, New York.Water Resources Bulletin 18:797-805.

Snover, A. K., A. F. Hamlet, and D. P. Lettenmaier. 2003. Climate-change scenarios for water planning studies - Pilot applications in the Pacific Northwest. Bulletin of the American Meteorological Society 84:1513-1518.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate 18:1136-1155.

United States Geological Survey. 2010a. Provisional data for Washington.

United States Geological Survey. 2010b. USGS Washington Water Science Center.

Vano, J. A., N. Voisin, L. Cuo, A. F. Hamlet, M. M. Elsner, R. N. Palmer, A. Polebitski, and D. P. Lettenmaier. 2009. Climate Change Impacts on Water Management in the Puget Sound Region, Washington, USA. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Wiley, M. W., and R. N. Palmer. 2008. Estimating the impacts and uncertainty of climate change on a municipal water supply system. Journal of Water Resources Planning and Management-Asce 134:239-246.

Violations of Instream Flow Rules

Background

Human demands for freshwater resources need to be balanced with the ecological needs of river and estuarine systems (Petts 2009). Instream flow rules, which allocate specific flow and timing regimes in rivers and river system, are meant to legally account for the ecological requirements that may have previously been unconsidered. The Washington Department of Ecology (Ecology) and Department of Fish and Wildlife (WDFW) have developed instream flow rules to "protect and preserve instream resources" (Washington State Department of Ecology 2004), that include fish and fish habitats, water quality, wildlife, aesthetics, and recreation. Instream flow rules are developed by a defined scientific methodology (Washington State Department of Ecology 2003). They do not affect established (senior) water rights or withdrawals. They can limit future (junior) surface water withdrawals, or withdrawals from ground water that is in hydraulic continuity with the surface water, in order to protect minimum instream flows. Instream flow rules may also limit maximum withdrawals or establish closures where it has been determined that there is no water available for further appropriations.

Instream flow rules do not affect exempt groundwater withdrawals, including:

- Stockwatering;
- Single or group domestic, up to 5,000 gallons per day;
- Industrial purposes, up to 5,000 gallons per day; and
- Irrigation of up to one-half acre of lawn or non-commercial garden (see Revised Code of Washington [RCW] 90.44.050).

Instream flow rules exist for many of the rivers and streams within the Puget Sound. Table 1 shows a summary of Instream Flow Rules for basins surrounding the Puget Sound by Water Resources Inventory Area (WRIA).

Alterations of the natural flow regime affect river ecosystems by changing physical habitats, including patterns of longitudinal and lateral connectivity, and by altering the natural cues and patterns of biological response, which could adversely affect native species that have evolved in response to historical flow patterns. Alterations could enhance the success of invasive or introduced species in a river system (Bunn and Arthington 2002). Due to the complexity of natural flow regimes, the establishment of simplified instream flow rules based on minimum flow requirements or rules of thumb may not be protective of natural resources; i.e., it is not clear whether instream flow rules are protective of native flora and fauna (Arthington et al. 2006, Naiman et al. 2008). Several studies have suggested the adoption of flow rules and management targets that are more considerate of all aspects of the natural flow regime (Bunn and Arthington 2002, Arthington et al. 2006, Naiman et al. 2008, Petts 2009, Poff et al. 2010).

A measure of the management effectiveness of freshwater resources is to compare actual instream flows with the instream flow rules. A high percentage of instream flow rule violations could indicate an over-allocation of freshwater in a basin. An increasing trend in violations could indicate that the freshwater demands are increasing. For the purposes of this report, violations were determined by comparing the instream flow rules to the average daily flow at specified gauging stations. A violation was noted when the average daily flow was less than that specified in the instream flow rule. The average percent of violation days per month were calculated for the time period of the instream flow rule. Trends were evaluated for the period from October to June or during the typically water-critical period from July toSeptember (see Table 2). Trends were determined by simple linear regression over time; trends significantly different that zero ($P<0.05$) were noted..

Violations for instream flow rules were calculated for eight rivers, with the intent of evaluating at least one river or stream from each of the WRIAs in the Puget Sound watershed. The selection of rivers is shown in Table 1.

Table 1. Summary of Instream Flow Rules for Water Resource Inventory Areas (WRIA) surrounding the Puget Sound.

Water Resources Inventory Area	Instream Flow Rule	Date	Closures
WRIA 1 - Nooksack	173-501 WAC	12/4/85	Yes
WRIA 2 - San Juan	No		
WRIA 3/4 - Lower Skagit-Samish and Upper Skagit	173-503 WAC ,	4/14/01, Update 6/15/06	No
WRIA 5 - Stillaguamish	173-505 WAC	9/26/05	Yes
WRIA 6 - Island	No		
WRIA 7 - Snohomish	173-507 WAC	9/6/79	Yes
WRIA 8 - Cedar-Sammamish	173-508 WAC	9/6/79	Yes
WRIA 9 - Duwamish-Green	173-509 WAC	6/6/80	Yes
WRIA 10 - Puyallup-White	173-510 WAC	3/21/80	Yes
WRIA 11 - Nisqually	173-511 WAC	2/2/81	Yes
WRIA 12 - Chambers-Clover	173-512 WAC	12/12/79	Yes
WRIA 13 - Deschutes	173-513 WAC	6/24/80	Yes
WRIA 14a - Kennedy-Goldsborough	173-514 WAC	1/23/84	Yes
WRIA 15 - Kitsap	173-515 WAC	7/24/81	Yes
WRIA 16/14b - Skokomish-Dosewalips	No		
WRIA 17 - Quilcene-Snow	173-517 WAC	12/31/09	Yes
WRIA 18 - Elwha-Dungeness	No		

Status and Trends

None of the river systems evaluated consistently met the instream flow rules (Table 2). In five of the eight river systems, there were at least two months per year when actual flows did not meet the instream flow requirements at least 50% of the time. Flows in the Stillaguamish River failed to meet instream flow rule requirements 90% of the time during the July-August-September period. This is the highest percent of violation of any river evaluated.

Table 2. Summary of percent violations of Instream Flow Rule for selected rivers in the Puget Sound. Period is effective dates of Instream Flow Rule. Violations occurred when average daily flow at gauging station was less than value specified by Instream Flow Rule. Overall average for the time period and annual percent change are shown. Water Resources Inventory Areas that are not shown do not have established Instream Flow Rules.

Water Resources Inventory Area	**USGS Gauge Station ID**	**Period**	**Average (Trend per year); %Violations per year[9]**	
			Oct – June	**July-Aug-Sept**
WRIA 1 - Nooksack	USGS 12213100 Nooksack River at Ferndale, WA	1986-2009	34 (- 0.3)	72 (- 0.3)
WRIA 3/4 - Skagit	USGS 12200500 Skagit River near Mount Vernon, WA	2002-2009	24 (+ 1.8)	54 (- 0.7)
WRIA 5 - Stillaguamish	USGS 12167000 NF Stillaguamish River near Arlington, WA	2005-2009	20 (- 1.5)	90 (- 2.5)
WRIA 7 - Snohomish	USGS 12144500 Snoqualmie River near Snoqualmie, WA	1979-2009	23 (+ 0.1)	53 (+ 1.3)
WRIA 8 - Cedar-Sammamish	USGS 12119000 Cedar River at Renton, WA	1979-2009	16 (- 0.3)	21 (- 0.9)
WRIA 9 - Duwamish-Green	USGS 12113000 Green River near Auburn, WA	1980-2009	15 (- 0.3)	55 (- 0.3)
WRIA 10 - Puyallup-White	USGS 12101500 Puyallup River at Puyallup, WA	1980-2009	7 (- 0.3)	10 (- 0.6)
WRIA 11 - Nisqually	USGS 12082500 Nisqually River near National, WA	1981-2009	23 (- 0.3)	21 (- 0.0)
WRIA 12 - Chambers-Clover[10]				
WRIA 13 - Deschutes[11]				
WRIA 14a - Kennedy-Goldsborough[12]				
WRIA 15 - Kitsap[13]				
WRIA 17 - Quilcene-Snow[14]				

Generally, the highest percent of violation of instream flow rules occurred in August and September . There were no significant trends of the percent violations of the instream flow rule over time for any of the river systems evaluated (P>0.05).

Uncertainties

This analysis uses average daily discharge data from the eight USGS stations specified in Table 2 (United States Geological Survey 2010b). The datasets include qualification codes indicating whether data are provisional or have been approved (United States Geological Survey 2010a). We avoided using provisional data in this analysis, and we omitted data from gauging stations for which advisory notes warning against unreliable data quality had been posted.The gauging stations on the NF Stillaguamish River near Arlington (USGS 12082500) and the Nisqually

River near National (USGS 12082500) advised of poor data quality during storms or high flow conditions. High flow conditions would not result in violations of the instream flow rules and so this did not affect the analysis.

The development and application of Instream Flow Rules is relatively recent (see Table 1). Consequently, most stations offer only a limited number of years from which to evaluate data. The relatively short time period and high interannual variability precluded detection of significant long term trends.

Summary

All streams showed violations of the instream flow rules, most commonly occurring in August and September. Notably, flow levels in the Stillaguamish River were below instream flow requirements approximately 90% of the time during the summer months. The Puyallup River exhibited the lowest percent of instream flow rule violations of any river evaluated. The monthly average percent violations did not exceed 25% for any month of the water year.

Literature Cited

Arthington, A. H., S. E. Bunn, N. L. Poff, and R. J. Naiman. 2006. The challenge of providing environmental flow rules to sustain river ecosystems. Ecological Applications 16:1311-1318.

Bunn, S. E. and A. H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30:492-507.

Naiman, R. J., J. J. Latterell, N. E. Pettit, and J. D. Olden. 2008. Flow variability and the biophysical vitality of river systems. Comptes Rendus Geoscience 340:629-643.

Petts, G. E. 2009. Instream Flow Science For Sustainable River Management. Journal of the American Water Resources Association 45:1071-1086.

Poff, N. L., B. D. Richter, A. H. Arthington, S. E. Bunn, R. J. Naiman, E. Kendy, M. Acreman, C. Apse, B. P. Bledsoe, M. C. Freeman, J. Henriksen, R. B. Jacobson, J. G. Kennen, D. M. Merritt, J. H. O'Keeffe, J. D. Olden, K. Rogers, R. E. Tharme, and A. Warner. 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. Freshwater Biology 55:147-170.

United States Geological Survey. 2010a. Provisional data for Washington.

United States Geological Survey. 2010b. USGS Washington Water Science Center.

Washington State Department of Ecology. 2003. A Guide to Instream Flow Setting in Washington State. Water Resources Program, Olympia, WA.

Washington State Department of Ecology. 2004. Setting instream flows and allocating water for future out-of-stream uses. Olympia, WA.